

ual, June 1986) of ± 0.11 ft ($\pm 0.5\%$ of full scale). The absolute accuracy depends on the accuracy of the setup. The nominal internal precision of these devices is listed as ± 0.003 ft, but the data printout (resolution) precision is usually to a level of ± 0.01 ft.

In another experiment conducted at a semi-arid site with water levels over 400 ft below the surface, Atwood and Lamb [2] reported a precision of ± 0.05 ft when two operators were using the same instrument at the same location within a short time period. A comparison of electric sounder measurements with those taken by a downhole transducer showed an overall uncertainty of ± 0.10 ft, which is generally attributed to sounder sensitivity and operator error.

Float-activated, continuous recorders have been in use for some time. According to Frank Riley (USGS) [1], the Stevens Type F recorder is the most accurate and simplest to use for continuous water level measurements. He reports that the overall accuracy achievable with this type of recorder is ± 0.001 ft. However, depth, borehole deviation, well diameter, and well access difficulties limit the usefulness of this device. Riley has noted that he has had good results in wells of 4-in. (and greater) diameter to depths of 200 to 300 ft. Special designs have been developed to use these recorders in wells of smaller diameter [3].

Jacob [4] was interested in the precision of a Stevens Type-F recorder while using different gage-to-height ratios. His initially recorded data employed a gage-to-height ratio of 1 to 1. For his more detailed studies, he modified the recorder to expand the time and water level scales and obtained more refined readings at a gage-to-height ratio of 5.1 to 1. This did, however, require a smaller float, which was observed to dampen the readings (i.e., the float lag was inversely proportional to the diameter of the float wheel). This effectively reduced the average recorded fluctuation from 0.03 to 0.016 ft. This information is emphasized in order to point out that when the causes of data fluctuation are identified, it may be possible to correct for the apparent error. Sources of type-F recorder error are discussed in more detail in the Leupold and Stevens data book [5].

Transducers

Currently, transducers are the instrument of choice for most continuous water level measurement. Riley [1] noted that he used transducers 30 years ago when battery life was a significant limitation on the usefulness of transducers. Modern electronic circuitry, better power sources, and the coupling of data loggers to computer systems for easy data reduction have greatly enhanced the value of these instruments. One shortcoming of transducer units is that they measure relative pressure or depth. The absolute or initial depth-to-water measurement is generally taken with a conductive probe or steel tape, thus limiting the absolute accuracy of the transducer/data logger to that of the pre-start calibration in most applications. Transducer types include the diaphragm transducer plus newer designs made up of bonded strain gage, vibrating wire, or vibrating crystal units.

When using transducers/data loggers, the precision of both units must be considered. The *Ground Water Monitoring Review* for summer 1987 [6] lists the types and manufacturers' claims of resolution or precision for a variety of transducers. This information has been corroborated by interviews with manufacturers' representatives. Achievable precision is dependent upon the scale required and the cost of the unit, and one should note that most manufacturers of transducers assume that the transducers are in an isothermal environment. The precision of transducers ranges from 0.25 to 0.10% of the full scale (FS). Transducers are commonly available in the 2 to 50-psi range. At 2.31 ft/psi, the FS range is 4.6 to 116 ft for the aforementioned transducers, with precision ranges from 0.01 to 0.005 ft for the 2-psi transducer and 0.29 to 0.12 ft for the 50-psi transducer.

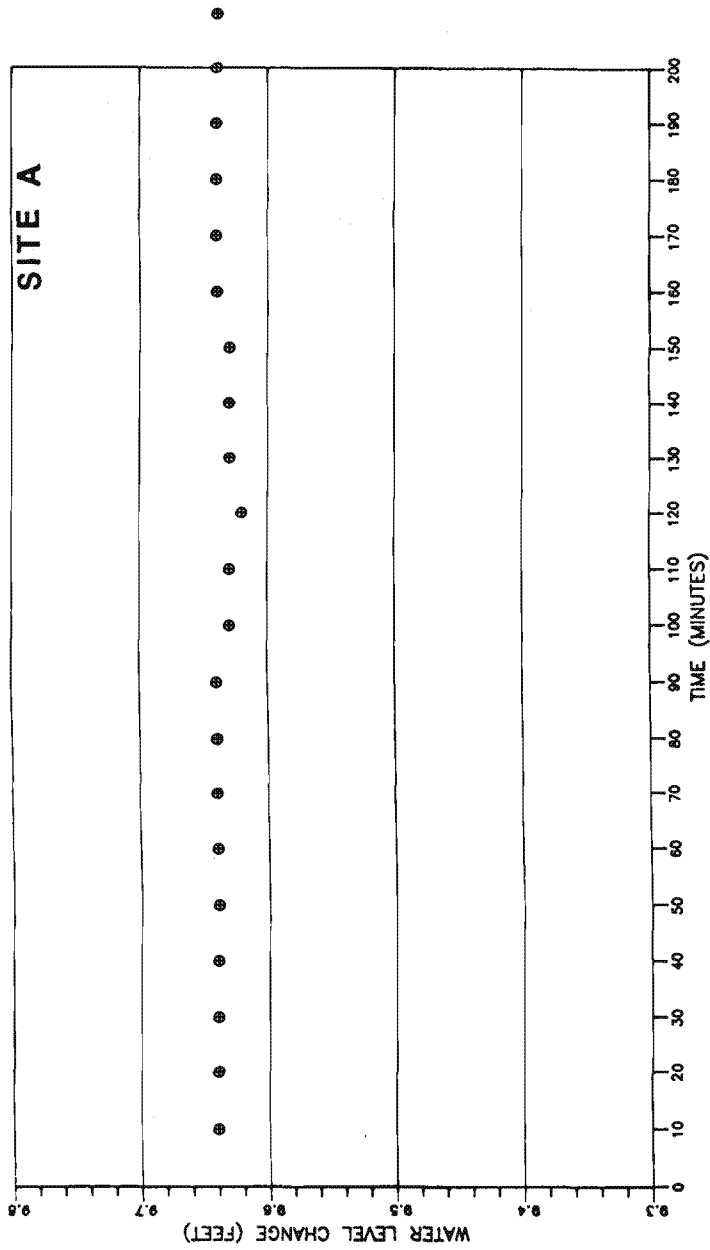


FIG. 1—Water level variations.

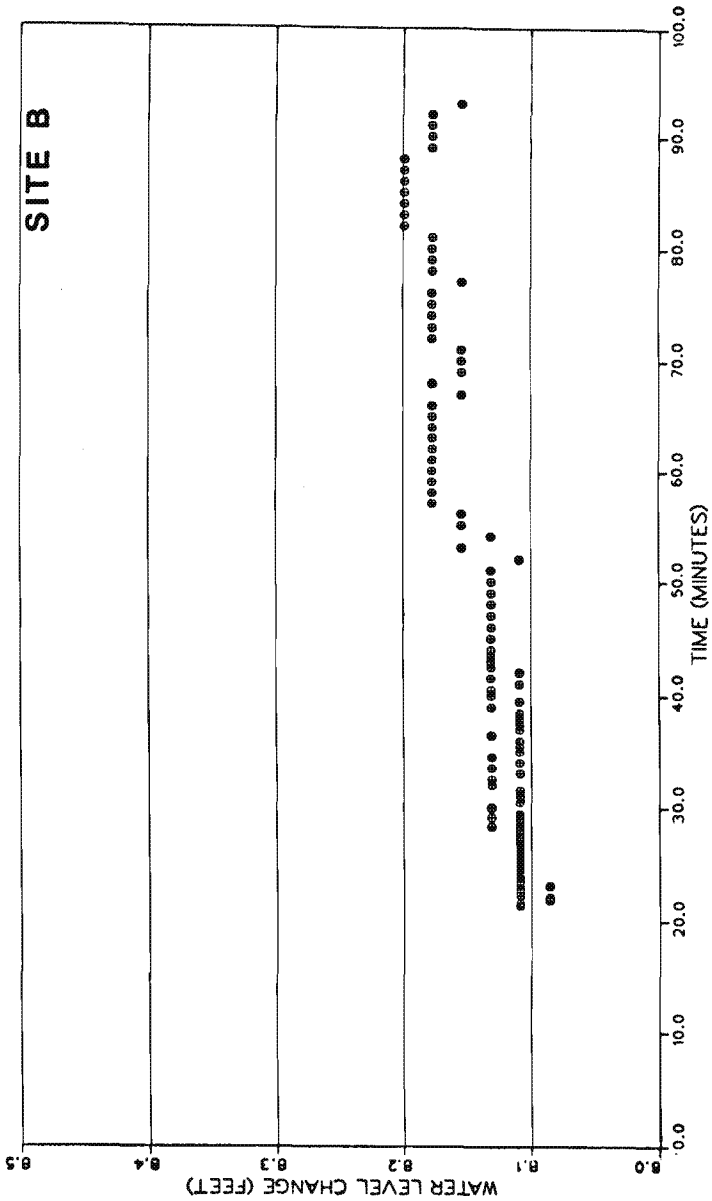


FIG. 1—Continued.

Transducers are commonly coupled to data logger units. If not properly paired, the resolution or readability of the data logger can be a limiting factor in the precision of the transducer/data logger system. As an example, the Terra 8 (Northwest Instrumentation) is capable of reading 1 unit/1024. For a 2-psi transducer with a 4.6-ft FS, the data logger precision is 0.004 ft (i.e., 4.6 ft/1024). Since the precision range for the 2-psi transducer is 0.01 to 0.005 ft, the data logger is adequate for this application. The Fig. 1 transducer plot for Site B illustrates the scale steps for a 10-psi transducer with 0.10% precision, or a plotting resolution of 0.02 ft.

Water Level Fluctuations—Natural

Another source of random accuracy problems is short-term water level fluctuation. As early as 1906, Veatch [7] compiled a comprehensive listing of water level fluctuation sources. In his listing he identified the following general categories: fluctuations due to natural causes and those due to "human agencies." Natural causes include rainfall, evaporation, barometric and temperature changes, rivers, ocean level changes (tides), and direct tides from solar or lunar attraction. Human causes included settlement, deforestation, cultivation, irrigation, dams, water supply developments, cities, and freight trains. He noted that many of the causes of fluctuations, such as precipitation, rivers, and tidal changes, are not due to an actual change in ground water storage but to a change induced by plastic deformation caused by the loading and unloading of the aquifer. Ferris et al. [8] expanded the list of causes and summarized more recent work in explaining effects. Freeze and Cherry [9] have also tabulated and discussed mechanisms that lead to fluctuations in ground water levels. As an example of natural fluctuations, in the cases of the two test experiments shown on Fig. 1, the water levels varied by 0.03 ft in 1000 min (0.3%) at Site A and 0.081 ft (0.99%) in 80 min at Site B.

A case can be made for categorization of barometric or diurnal water level fluctuations as random in short-term, or systematic in longer term, experiments. Making corrections for barometric pressure changes during long-term pump tests has been a standard practice since the early 1940s [10]. In order to emphasize the time dependency as well as the magnitude of natural water level fluctuations in relatively deep aquifers in a semi-arid region, Atwood and Lamb [2] placed downhole transducers in two wells, A and B, completed at different water-bearing zones for a two-month period. Two superimposed water level fluctuation patterns were observed in both wells (See Figs. 2 and 3). The first pattern observed is cyclic and diurnal with average fluctuations of ± 0.20 ft. The second pattern has an irregular cycle which spans two to seven days with fluctuations averaging ± 0.1 ft. However, several larger fluctuations exceeded 0.5 ft, with a maximum of 0.8 ft. These large changes in water levels were interpreted as being caused by barometric response to passing weather fronts. These responses are temporally synchronized (Figs. 2 and 3) in the wells, but the magnitude of the fluctuation is different in each well. Figure 4 illustrates natural and diurnal fluctuation interpretations.

Water Level Fluctuations—Anthropogenic

Random anthropogenic fluctuations of water levels are also well documented. In 1939, Jacob [4] wrote a now-classic paper on the water level changes induced by a passing train. The magnitude of fluctuations that he observed ranged from 0.024 to 0.035 ft and averaged 0.030 ft. The differing response depended primarily on the weight and velocity of the train, with increased weight and velocity increasing the magnitude of the water level change.

During site characterization and testing of a contaminant capture system at a RCRA site

in western Oregon, an analogous effect was noted in a shallow semiconfined aquifer. At this site, transducer/data logger sets were installed in seven monitoring wells and piezometers surrounding a contaminant capture well. Figure 5 shows the drawdown response in one observation well over a 15-day period. What initially appeared to be a random fluctuation in the water levels recorded by the transducer/data logger sets was later interpreted to be a systematic change, apparently driven by diurnal parking lot loading and unloading over the shallow but partially confined aquifer. The tightly packed data set, therefore, allowed an interpretation and correction of the long-term measured and projected drawdown.

Special Problems

At highly contaminated facilities, special problems arise when attempting to measure water levels. Multiphasic liquids (e.g., floating oil in irrigation wells from deep turbine pumps) have long posed a problem in measuring true water levels. It is not uncommon to encounter several feet of accumulated oil in such a system. Light nonaqueous-phase liquids, such as those from petroleum tank leaks, have become a common problem in many underground storage tank (UST) investigations. Experience has shown that the use of petroleum paste on a steel tape can provide a measurement accuracy of ± 0.10 ft. Clear acrylic bailers used in the measurement and collection of floating products can measure from a film or sheen of 0.001 ft (in shallow floating plumes) to ± 0.10 ft (in thicker plume layers).

Another special case encountered in landfill monitoring wells is the interference resulting from foaming leachate and methane gas pressures. Water and leachate levels, as well as foam generation in highly contaminated wells, are dependent upon the rate of gas generation, gas pressures built up within the waste, the barometric pressure confining that gas at any given point in time, or a combination of these factors. Our experience has shown that the measurements within these landfill wells can be virtually impossible to obtain with either conductive or transducer devices and may “naturally” vary as much as 10 ft in a matter of hours.

Systematic Errors—Well Problems

Systematic errors are primarily anthropogenic and include surveying accuracy, well deviations, instrument deterioration (e.g., cable stretching), and poor calibration. Wellhead surveying problems have been discussed by Atwood and Lamb [2], with a reported accuracy resolution of 0.01 ft. As discussed earlier, cable stretch data are not generally available but may be on the order of 0.02 to 0.10%. The calibration and care of the instruments are operator dependent and very difficult to quantify.

Well deviation, especially in deeper monitoring systems can be a major concern. At a semi-arid site in California's Central Valley, this was studied in some detail. The site is underlain by dipping water-bearing sandstone units which are confined between low permeability claystones. The depth to ground water is typically greater than 400 ft from the surface. The hydraulic gradient in the water-bearing sandstones is very small, averaging less than 1 ft over 1000 ft. With such a shallow gradient, it was imperative that the highest possible accuracy be maintained in order to define ground water movement. Natural water level fluctuations were evaluated at the site and have been discussed earlier. Well deviation from the vertical was suspected as a major element of inaccuracy. In a well which is not exactly vertical, the actual water depths are shorter than those measured with a conductance probe or other measuring instrument. Minor well deviation occurs in most site wells

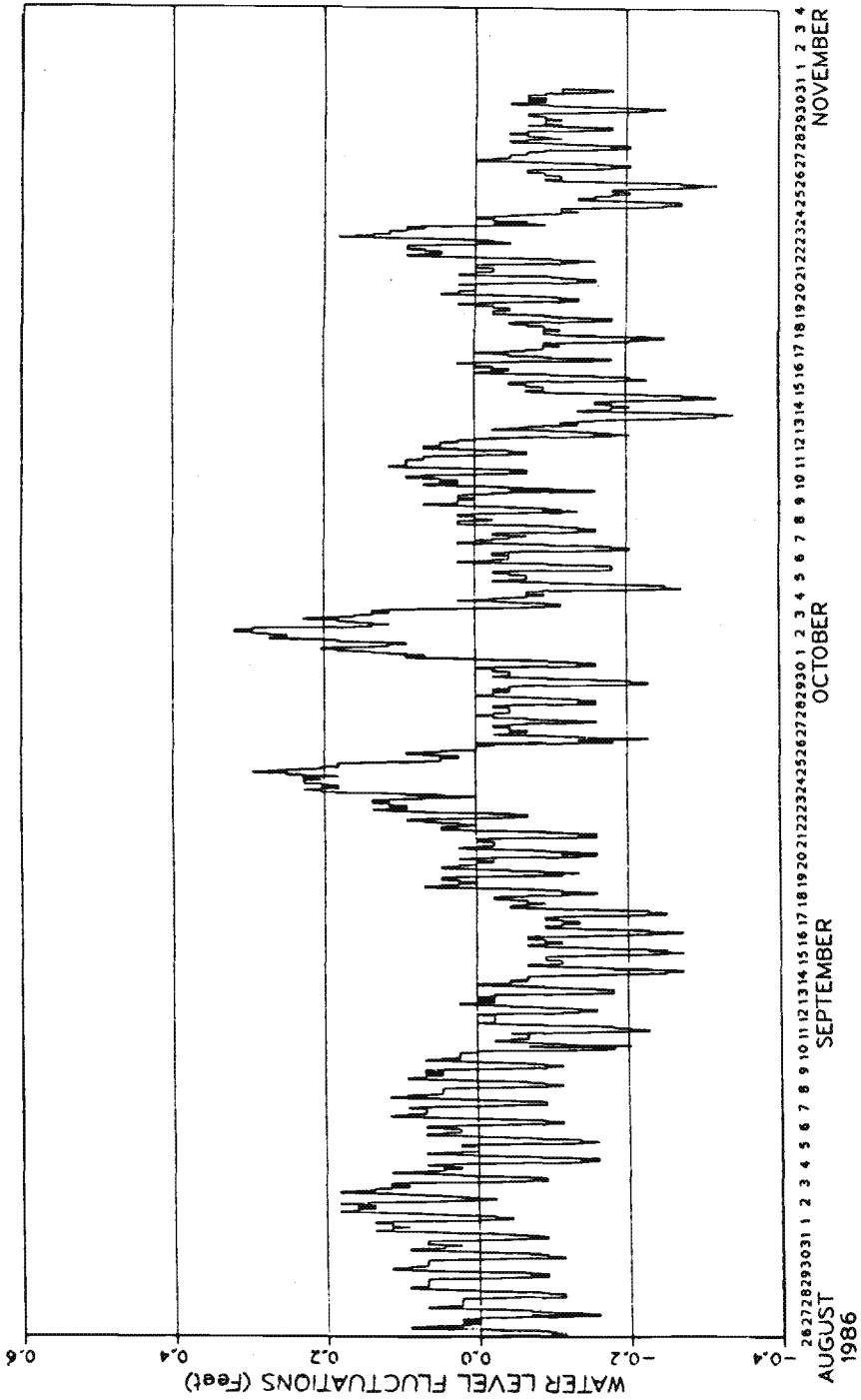


FIG. 2—Water level fluctuations in Well A.

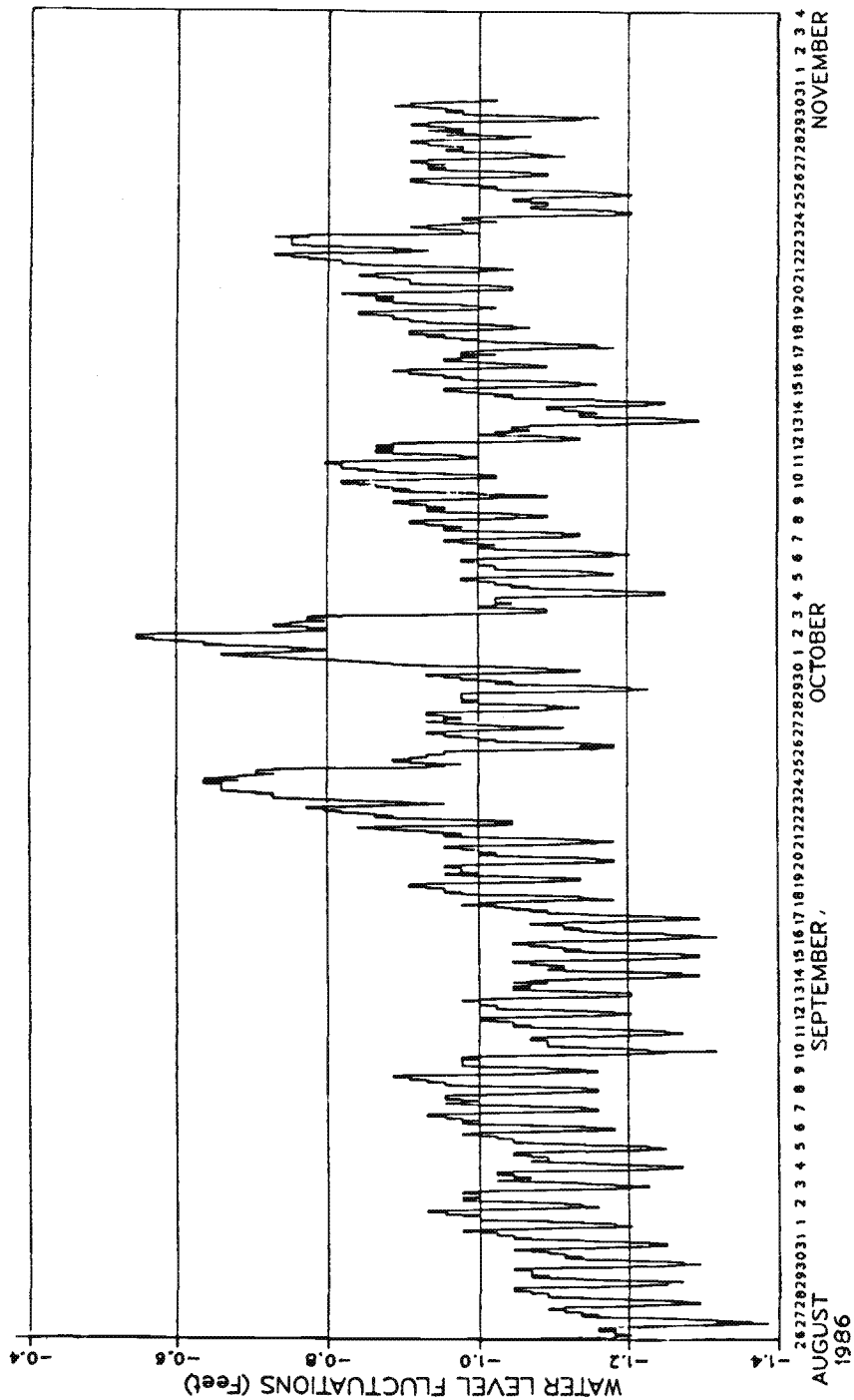


FIG. 3—Water level fluctuations in Well B.

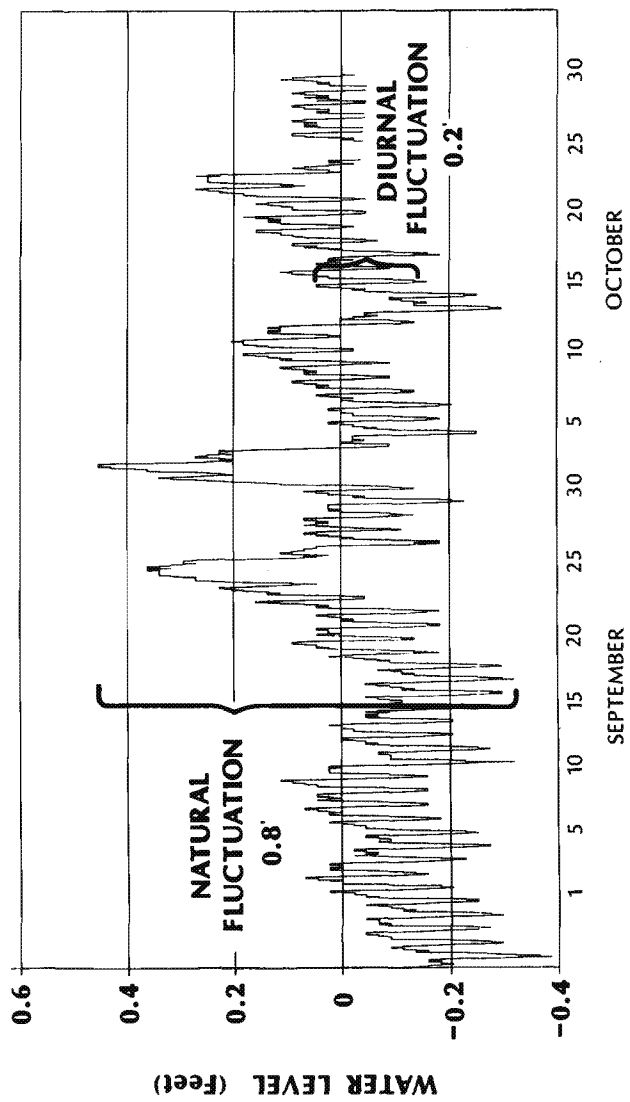


FIG. 4—Natural versus diurnal fluctuations.

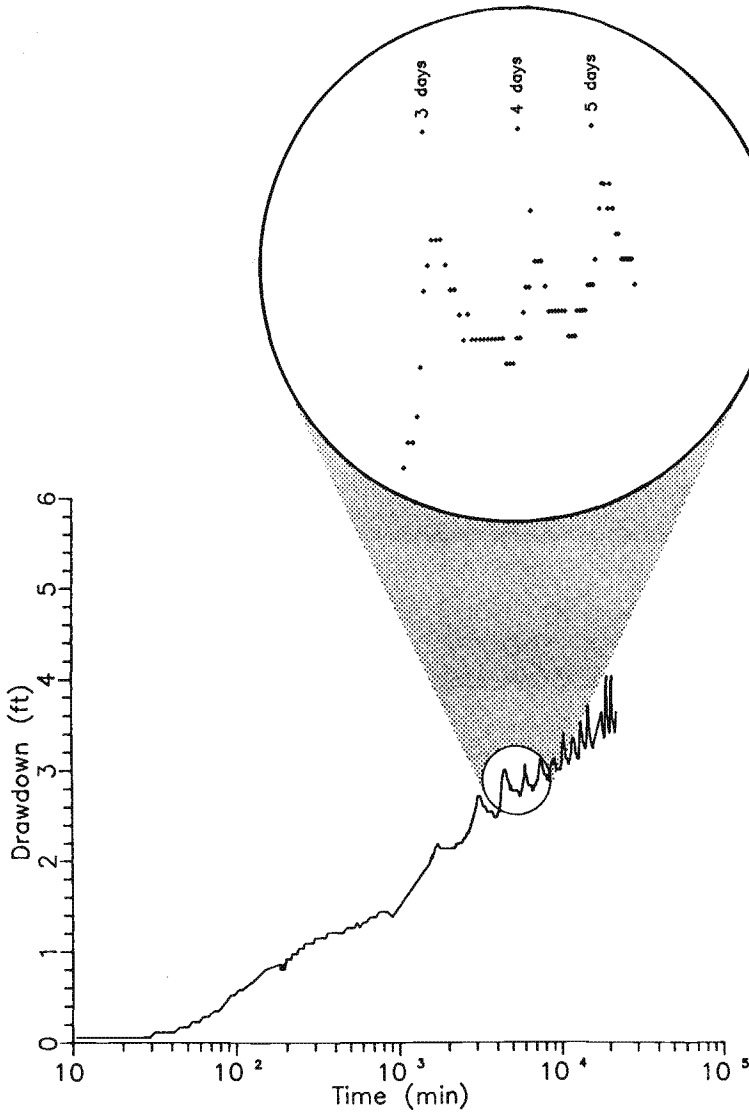


FIG. 5—Drawdown response with parking lot loading and unloading.

because of the great depth of the wells and the dipping strata, which cause the boreholes to generally drift into the dipping beds.

Over the length of these wells, the error introduced by deviation can be significant. Fortunately, this type of error can be corrected by measuring actual deviation with one of several available downhole instruments. The borehole drift tool was used most frequently during this site investigation. This downhole probe contains a transducer for sensing degree and direction of inclination. The magnetic-multishot survey is another technique used for deviation determination and consists of a 1.75-in.-diameter probe which records a compass and inclinometer reading on a film as the probe is lowered down the hole. A gyroscopic survey, the third type utilized during the site investigation, was employed in wells with steel casing materials. This 1.75-in.-diameter probe has a spin motor which is aligned to a surface reference of known orientation. Readings are then recorded on film as the probe is lowered down the hole.

Deviation from true vertical increases with depth. In this particular site investigation, the vertical correction at the piezometric surface ranged from -0.01 to -5.97 ft. If uncorrected, this systematic error could completely obscure the true gradients between adjacent wells.

A deviation survey corrects for a well's variation from the vertical. However, the probes and instruments used to measure the well deviation have a limited precision. The survey companies have reported the precision of the equipment and survey technique to be ± 0.30 ft at this site. Even when corrected, the precision of these instruments creates a systematic precision error that limits overall water level measurement accuracy.

Cumulative Error

Random errors are generally not cumulative, but can lead to large uncertainties. Systematic errors may or may not accumulate to produce even greater inaccuracies. An example of opposing systematic errors is the combined effects of uncorrected well deviation and measurement device stretching. Cumulative errors may combine anthropogenic factors, weather effects, and slow temporal variations. An important principle is that while inaccuracies may cancel, they are often not predictable, while uncertainties are generally additive. That is, the uncertainty in any set of calculations will be larger than the poorest measurement unless the number of measurements increases to compensate. This is particularly a problem when an aquifer parameter is calculated from a set of variables or parameters, each of which, like a depth to water measurement, is subject to its own characteristics of inaccuracy and imprecision. Calculations using precise values for some factors (e.g., gradients) will be limited by the imprecisions of other factors.

For calculated values using additive or subtractive values, the uncertainties are approximately additive, while for multiplicative factors, the percentage errors are roughly additive. As an example, in situations involving very shallow gradients, additive uncertainties in water levels may lead to percentage uncertainties in the gradient of a similar magnitude (e.g., $\pm 50\%$) to hydraulic conductivity uncertainty. In such cases, the 0.1 to 0.25% uncertainties or inaccuracies (0.1 to 0.25 ft/100 ft), which are nearly impossible to avoid in water level measurement, may result in $\pm 100\%$ uncertainties in calculated flow velocities.

Conclusions

Table 1 summarizes instrument precision data from various manufacturers' literature. These range from ± 0.005 ft/100 ft (0.005%) to ± 0.25 ft/100 ft (0.25%). Conductive probe data from shallow well measurements (Sources 3 and 4 in Table 1) represent true mea-

TABLE 1—Water level measurement—relative precision data.

Method	Source ^a	Precision, %
Steel tape	(1)	0.005
Conductive probe	(2)	0.04
	(3)	0.21
	(4)	0.34
	(5)	0.01
Transducer	(6)	0.25
	(7)	0.10
	(8)	0.25
	(9)	0.10
	(10)	0.01

^a Sources:

- (1) U.S. Geological Survey (Frank Riley, personal communication, 1987).
- (2) Manufacturers' literature and interviews.
- (3) Shallow water table experiment, Location A.
- (4) Shallow water table experiment, Location B.
- (5) Atwood and Lamb (1987), deep wells.
- (6) 2 psi at 0.25%.
- (7) 2 psi at 0.10%.
- (8) 50 psi at 0.25%.
- (9) 50 psi at 0.10%.
- (10) 1 unit/1024 as per the Terra 8 specification.

surement precision and include operator plus instrument factors, with uncertainties of ± 0.21 to $\pm 0.34\%$. Source 5 in Table 1 is for a deep well experiment and includes operator plus instrument factors, at $\pm 0.01\%$ uncertainty. The percentage error decreases with depth since the operator precision is independent of depth. This implies a maximum precision of ± 0.02 ft attributable largely to operator error.

Accuracy goals must consider the limiting instrument and measurement precision as well as random and systematic sources of inaccuracy. As a rule of thumb, with few data points, accuracy is one order of magnitude (i.e., one decimal point to the left) lower than the limiting precision. As the measurement data population increases, accuracy limits approach precision limits.

This is not to imply that an infinite data base is necessary for all scientific investigations or projects. It does point out that the size of the data base should be designed in consideration of instrument and operation precision as they relate to a desired accuracy goal. As Cedergren [1] once said, "An approximate solution to the correct problem is far better than a refined solution to the wrong problem."

Acknowledgments

Although the mention of trade names and products does not constitute an endorsement, the authors are grateful to the various manufacturers' representatives for their information, cooperation, and candid comments. Staff in the various offices are thanked for assisting the experiments described herein.

References

- [1] Frank Riley, U.S. Geological Survey, Menlo Park, CA, personal communication, 1987.
- [2] Atwood, D. F. and Lamb, B., "Resolution Problems with Obtaining Accurate Ground Water

Elevation Measurements in a Hydrogeologic Site Investigation," *Proceedings*, First National Outdoor Action Conference on Aquifer Restoration, Ground Water Monitoring, and Geophysical Methods, National Water Well Association, May 1987, pp. 185-193.

- [3] Shuter, E. and Johnson, A. I., "Evaluation of Equipment for Measurement of Water Levels in Wells of Small Diameter," *U.S. Geological Survey Circular*, No. 453, 1961.
- [4] Jacob, C. E., "Fluctuations in Artesian Pressure Produced by Passing Railroad Trains, as Shown in a Well on Long Island, New York," *American Geophysical Union Transactions*, Part IV, 1939, pp. 666-674.
- [5] *Stevens Water Resources Data Book*, 2nd ed., Leupold and Stevens, Portland, OR, 1978.
- [6] *Ground Water Monitoring Review*, Vol. 6, No. 3, Summer 1987.
- [7] Veatch, A. C., "Fluctuations of the Water Level in Wells with Special Reference to Long Island, New York," Water Supply Paper 155, U.S. Geological Survey, Washington, DC, 1906.
- [8] Ferris, J. G., Knowles, D. B., Brown, R. H., and Stallman, R. W., "Theory of Aquifer Tests," Water Supply Paper 1536-E, U.S. Geological Survey, Denver, CO, 1962.
- [9] Freeze, R. A. and Cherry, J. A., *Groundwater*, Prentice-Hall, Englewood Cliffs, NJ, 1979.
- [10] Jacob, C. E., *American Geophysical Union*, Vol. 21, 1940, pp. 574-586.
- [11] Cedergren, H. R., "Seepage, Drainage and Flow Nets," Wiley, New York, 1968.

Presentation of Water Level Data

REFERENCE: Henning, R. J., "Presentation of Water Level Data," *Ground Water and Vadose Zone Monitoring, ASTM STP 1053*, D. M. Nielsen and A. I. Johnson, Eds., American Society for Testing and Materials, Philadelphia, 1990, pp. 193–209.

ABSTRACT: Most common methods used to present water-level data are graphical. These include various kinds of hydrographs, such as those showing water level versus time, draw-down versus time or distance, or the level in one well versus that in another. Contour plots in map view are essential for evaluating ground-water flow. Water levels on cross sections are useful in evaluating the relationship between stratigraphic units and water levels. Flow nets can be used in plan or cross-sectional view to estimate flow lines and possible pathways for seepage.

KEY WORDS: ground water, water level, data presentation, graphical presentation, tabular presentation

It is not enough for an investigator to collect water-level data. Once measurements have been made, they must be organized, evaluated, and interpreted [1]. As with many environmental measurements, those investigating frequently do not have a sampling framework for the populations under study. Some basic assumptions can be made, based on physical measurements of the system, but one is never sure that he understands fully the system he is attempting to quantify. For example, it is often believed that the ground surface is the hypothetical top of the ground water; however, numerous flowing artesian wells have pressure levels above the land surface regime even though the top of the physical water is below the ground at the aquitard. The water that rises in a cased well is the expression of the pressure of the aquifer.

The three broad objectives for gathering engineering data are to discover (a) the physical constants and frequency distributions, (b) the relationships—both functional and statistical—between two or more variables, and (c) the causes of observed phenomena [2]. Most techniques for collecting water-level data depend on three assumptions. First, we assume that water levels at a particular point are continuous in space. Inherent in this first assumption is the assumption that the point monitored represents actual conditions and bears a relationship to the water levels surrounding it in three dimensions. Second, we assume that the water level at a particular time is autocorrelated to previous water-level measurements; that is, some information is repeated or carried over from one observation to the next. Third, we assume that water-level populations have a statistical distribution and that we can sample representative values from a population distribution.

This paper will concentrate on the common presentation forms often used in the general literature. Because English units are still the standard in the United States, all data in the illustrations are presented exactly as originally collected.² Numerous statistical and prob-

¹ Chief hydrogeologist, Woodward-Clyde Consultants, Wayne, NJ 07470.

² Note that 1 ft = 0.3048 m, and 1 in. = 25.4 mm.

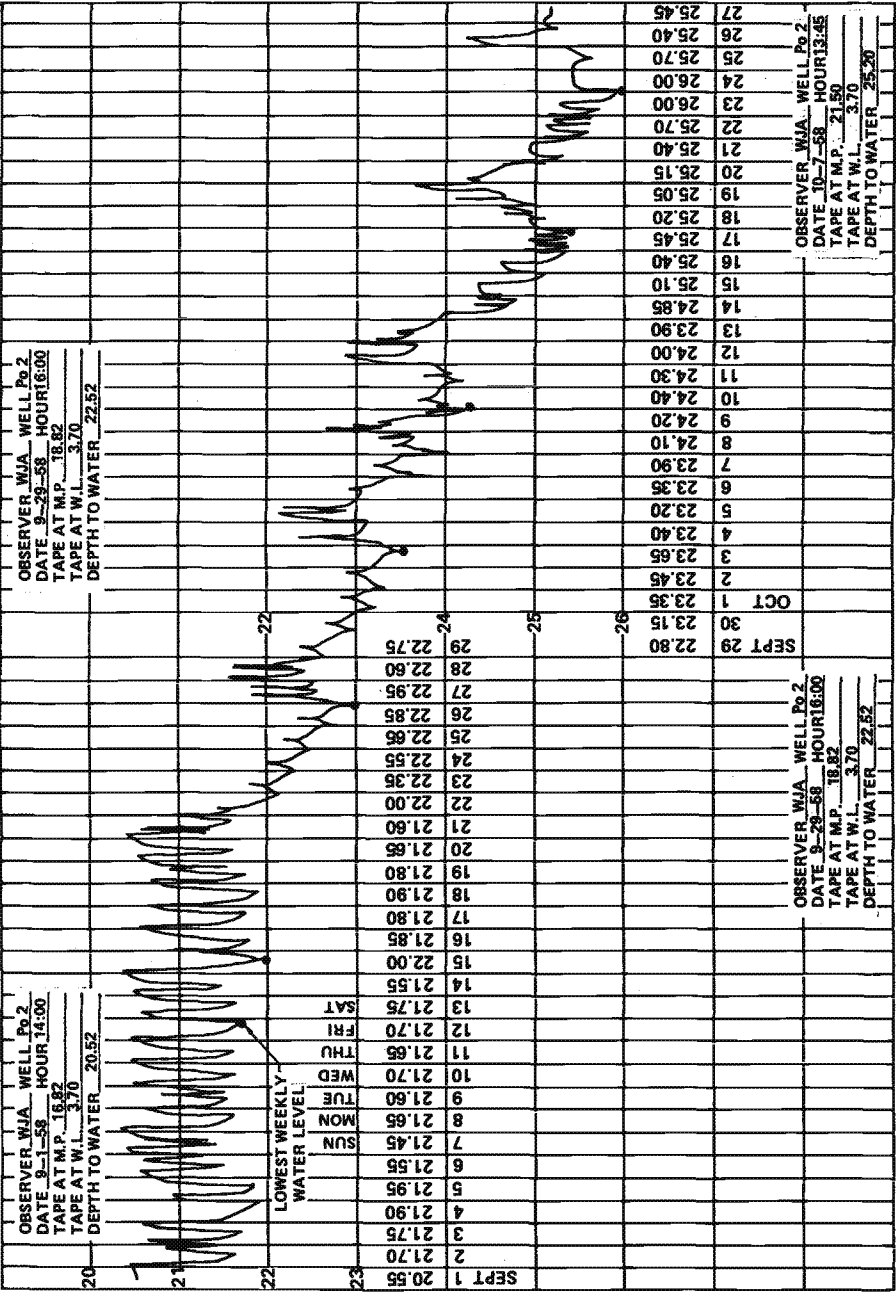


FIG. 1a—Monthly continuous recorder charts [3]. The measurements are in feet (1 ft = 0.3048 m).

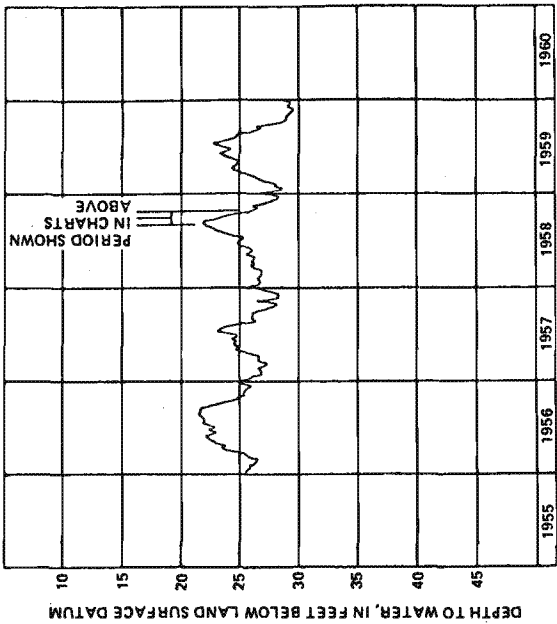


FIG. 1c—Portion of the ten-year-by-month hydrograph (the monthly lines are omitted) [3] (1 ft = 0.3048 m).

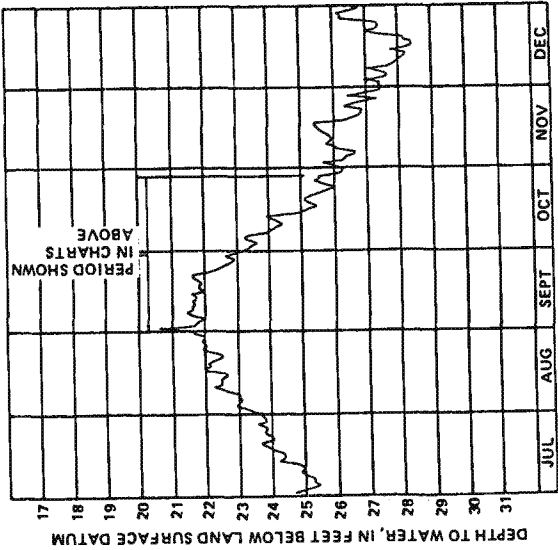


FIG. 1b—Portion of the year-by-day hydrograph (the daily lines are omitted) [3] (1 ft = 0.3048 m).

HUNTERDON COUNTY

402644074363601. Local I.D., Bird Obs. NJ-WRD Well Number, 19-0002.
LOCATION: Lat 40°26'44", long 74°56'36", Hydrologic Unit 02040103, at U.S. Post Office, Sergeantsville.
Owner: Phillip Fleming.
AQUIFER: Stockton Formation of Triassic age.
WELL CHARACTERISTICS: Dug water-table observation well, diameter 3 ft. (0.9 m), depth 21 ft. (6.4 m), lined with stone.
INSTRUMENTATION: Water level recorder.
DATUM: Land-surface datum is 342.00 ft. (104.242 m) above National Geodetic Vertical Datum of 1929.
Measuring point: top edge of recorder shelf, 1.50 ft. (0.460 m) above land-surface datum.
PERIOD OF RECORD: June 1965 to July 1970, May 1977 to current year. Periodic manual measurements, September 1970 to September 1976. Records for 1965 to 1976 are unpublished and are available in files of New Jersey District Office.
EXTREMES FOR PERIOD OF RECORD: Highest water level, 6.91 ft. (2.106 m) below land-surface datum, March 28-29, 1978 and April 2, 1980; lowest, 17.04 ft. (5.194 m) below land-surface datum, January 26-28, 1981.
EXTREMES FOR CURRENT YEAR: Highest water level, 10.11 ft. (3.082 m) below land-surface datum, January 9; lowest, 16.18 ft. (4.932 m) below land-surface datum, October 24.

WATER LEVEL, IN FEET BELOW LAND SURFACE DATUM,
WATER YEAR OCTOBER 1981 TO SEPTEMBER 1982
MEAN VALUES

DAY	OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP
5	15.65	15.08	14.36	-	-	13.43	10.90	11.80	13.95	14.25	14.69	15.19
10	15.63	15.30	14.22	10.59	-	11.98	10.86	13.07	14.70	14.62	15.24	15.34
15	15.91	15.70	-	12.17	-	12.32	10.55	13.93	12.64	14.43	14.96	15.72
20	16.14	15.51	-	13.28	-	12.58	11.13	14.37	12.47	15.04	15.38	15.90
25	16.01	15.56	-	13.84	12.46	13.17	12.17	14.35	13.72	15.29	15.61	15.48
EOM	14.67	15.89	-	-	12.86	13.85	10.75	13.85	14.49	13.89	15.60	15.03
MEAN	15.72	15.42	-	12.82	-	12.88	11.38	13.41	13.71	14.70	15.15	15.52
WTR YR 1982	MEAN 14.10	HIGH 10.23	JAN 9	LOW 16.17	OCT 23	AND OTHERS						

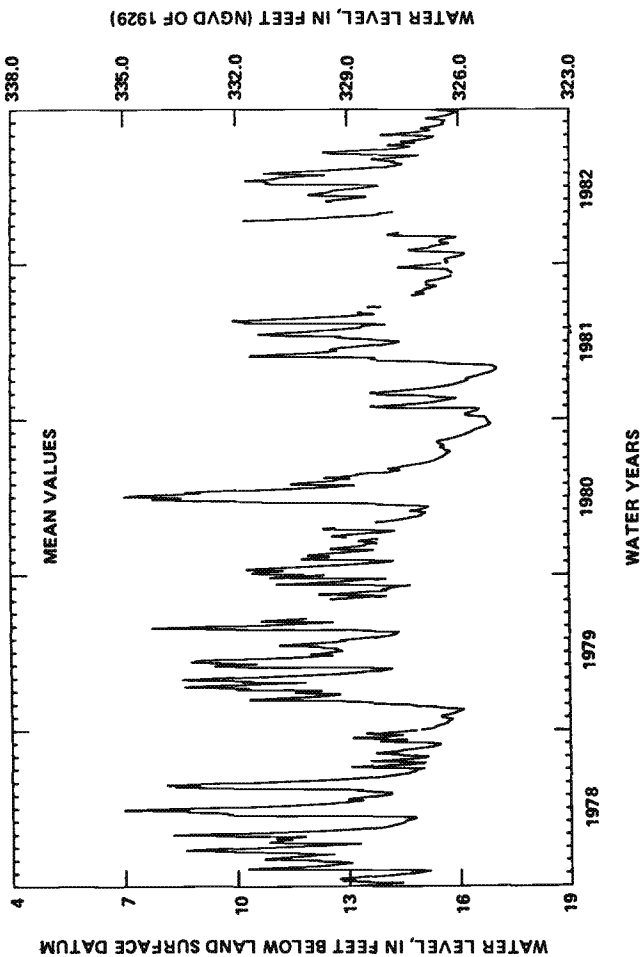


FIG. 2a—Water level data for WRD Well No. 19—0002, Hunterdon County, NJ [4] (1 ft = 0.3048 m; 1 in. = 25.4 mm).

OCEAN COUNTY

395714C74223401. Local I.D., Crammer Obs. NJ-WRD Well Number, 29-0486.
LOCATION: Lat 39°57'14"N, long 74°23'34"W; Hydrologic Unit 02040301, about 800 ft. (244 m) east of Central Railroad of New Jersey, Whiting.
Owner: Frank Reynolds.
AQUIFER: Kirkwood-Cohansey aquifer system of Miocene age.
WELL CHARACTERISTICS: Water-table observation well, diameter 8 in. (203 mm), depth 69 ft. (21.0 m), slotted steel casing, gravel packed.
INSTRUMENTATION: Water-level recorder.
DATUM: Land-surface datum is 179.00 ft. (54.559 m) above National Geodetic Vertical Datum of 1929.
Measuring point: top of 8-in. coupling, 090 ft. (0.270 m) above land-surface datum.
REMARKS: Originally a dug well which casing was inserted on March 31, 1966, and the well deepened from 60 to 69 ft. (18.3 to 21.0 m).
PERIOD OF RECORD: May 1952 to current year.
EXTREMES FOR PERIOD OF RECORD: Highest water level, 47.80 ft. (14.569 m) below land-surface datum, June 9-14, 20-29, 1973; lowest, 57.58 ft. (17.550 m) below land surface datum, June 1, 1966; well was dry, November 1957 to February 1958, and December 1965, before deepening.
EXTREMES FOR CURRENT YEAR: Highest water level, 56.40 ft. (17.191 m) below land-surface datum, July 13-17; lowest, 57.40 ft. (17.496 m) below land-surface datum, January 2-9.

WATER LEVEL, IN FEET BELOW LAND SURFACE DATUM, WATER YEAR OCTOBER 1981 TO SEPTEMBER 1982 MEAN VALUES											
DAY	OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG
5	57.06	57.24	57.31	57.40	57.28	57.04	56.82	56.76	56.70	56.45	56.50
10	57.09	57.25	57.34	57.39	57.24	57.00	56.81	56.76	56.66	56.41	56.52
15	57.11	57.26	57.35	57.38	57.19	56.96	56.78	56.77	56.61	56.40	56.54
20	57.15	57.27	57.37	57.36	57.15	56.91	56.76	56.79	56.56	56.41	56.58
25	57.19	57.29	57.38	57.34	57.11	56.88	56.76	56.81	56.51	56.46	56.62
EOM	57.23	57.31	57.39	57.31	57.09	56.85	56.77	56.75	56.48	56.50	56.67
MEAN	57.13	57.26	57.35	57.37	57.20	56.95	56.79	56.78	56.60	56.44	56.56
WTR YR 1962	MEAN	56.93	HIGH	56.40	JUL 14 AND OTHERS	LOW	57.40	JAN 2 AND OTHERS			

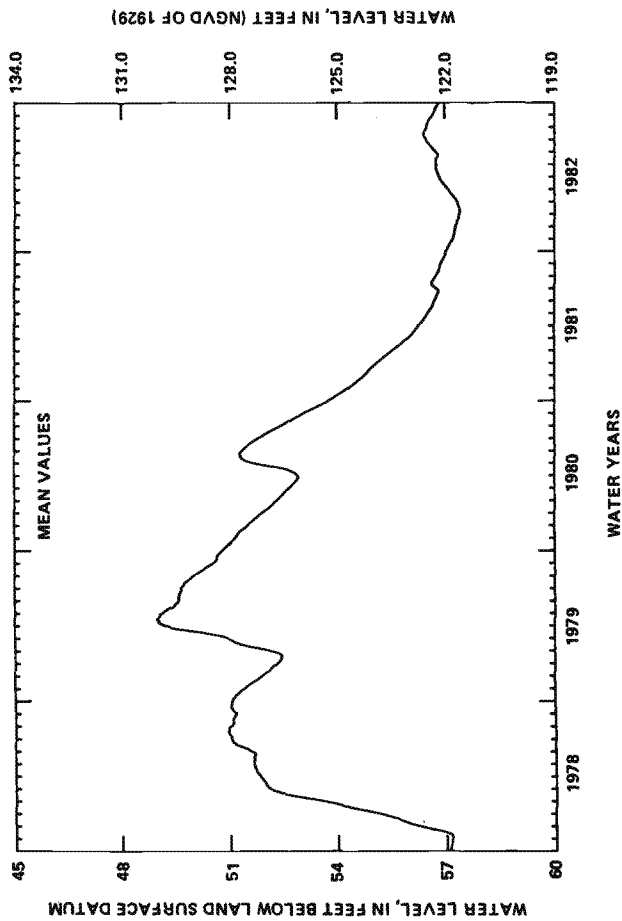


FIG. 2b—Water level data for WRD Well No. 29—0486, Ocean County, NJ [4] (1 ft = 0.3048 m; 1 in. = 25.4 mm).

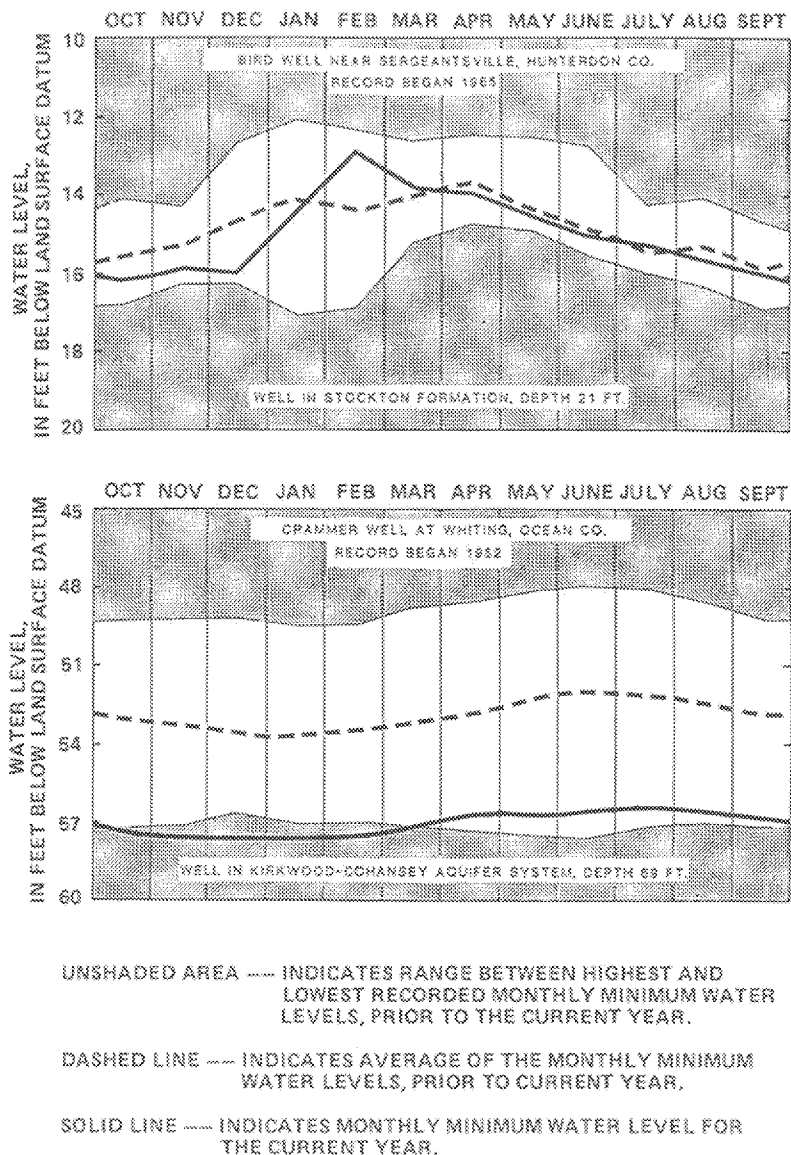


FIG. 2c—Monthly ground water levels at the two key observation wells in Figs. 2a and 2b [4] (1 ft = 0.3048 m).

ability graphical presentations have been used in the literature to interpret and present ground-water data, but these are beyond the scope of this paper.

We know from continuous water-level recorders that water levels are a continuous function in almost all natural aquifers and are controlled by boundaries or imposed stresses or both. Numerous studies and thousands of water-level records indicate that ground-water levels are probably normally distributed, especially in a dynamically stable natural system.

If the aquifer experiences stresses that cause boundary conditions to exert a control, then a bimodal distribution is likely to be produced. This is a result of the combination of the normal distribution and a random distribution reflecting the stress and boundary effects.

Accuracy and precision of measurements are important issues, but these are not discussed in this paper. The data presentation here assumes that the representative field data have been carefully collected and recorded in a consistent manner, converted, if necessary, to a standard engineering unit, and checked for accuracy and precision. The data set is then ready for processing into a working or final presentation form.

Hydrographs

Most of the common methods used to present water-level data are graphical. These include various kinds of hydrographs. Hydrographs are x - y plots of water-related data shown in relation to some other measurable continuous variable, such as time or distance. These graphs commonly depict the water level versus time, drawdown versus time or distance, or some calculated value, such as elevation above mean sea level (or some other convenient datum point). These simple plots allow the analyst to compare information visually. The graph can easily be examined to determine the minimum and maximum values and trends. Often, a trend is complex and may be easier to evaluate visually than to analyze mathematically.

The most common hydrograph is a continuous water-level recorder chart, illustrated in Fig. 1*a*. This figure illustrates copies of two successive monthly water-level recorder charts [3]. The initial quality assurance checks and manual digitization of the daily average water level, reported as a depth below a known reference point, have been completed. From these digitized data, we can show the average daily water level, illustrated in Fig. 1*b*. Filtering the data set again to remove monthly perturbations produces a long-term plot covering many years (Fig. 1*c*).

Combination presentations, such as those found in the U.S. Geological Survey (USGS) Water Resources Division (WRD) reports, present both text, tabular, and graphical information on the stations. Figure 2*a* illustrates the record of a dug well located in Hunterdon County, New Jersey, which responds quickly to recharge and probably to local discharge to streams [4]. The tabular results presented, although potentially more accurate, do not give the analyst the ability to infer time relationships. It is readily apparent from the graphical presentation that this well exhibits wide variations with time. It would probably not be appropriate to measure the water level monthly and expect the results to represent the true population.

Figure 2*b* illustrates a water-table well located in Ocean County, New Jersey, where the ground-water levels are much deeper. It is not dominated by short-term recharge and discharge events, but shows a more muted response consistent with time. The record for this well clearly shows that the drought, which began in late 1980, had a severe impact on water levels during 1981 and 1982. The monthly ground-water level data for both of these wells (Figs. 2*a* and 2*b*) are summarized in Fig. 2*c*, which combines period-of-record monthly minimums and maximums, with plots of monthly minimums for the previous year and the current year. Comparison of these two key observation wells shows that the drought caused the Cramer well in Ocean County to drop below historic lows from October through March of the water year 1982 (1982–83). However, the Bird well in Hunterdon County showed monthly minimums for both 1981 and 1982 that apparently did not differ appreciably from the historic record.

Figure 3 illustrates a presentation of data from an artesian aquifer that is instrumented

OCEAN COUNTY

395609074124001. Local I.D., Toms River TW 2 Obs. NJ-WRD Well Number, 29-0534.
LOCATION: Lat 39°56'09", long 74°12'14", Hydrologic Unit 02040301, about 200 ft. (61.0 m) east of Double Trouble Road on the north side of Jakes Branch, South Toms River. Owner: U.S. Geological Survey.
AQUIFER: Englishtown aquifer of Cretaceous age.
WELL CHARACTERISTICS: Drilled artesian observation well, diameter 8 in. (203 mm), depth 1146 ft. (349.3 m), screened 1080 to 1146 ft. (329.2 to 349.3 m).
INSTRUMENTATION: Water-level extremes recorder, February 1977 to current year. Water-level recorder, December 1965 to March 1975.
DATUM: Land-surface datum is 18.34 ft. (5.590 m) above National Geodetic Vertical Datum of 1929.
Measuring point: front edge of cutout in recorder housing, 1.70 ft. (0.518 m) above land-surface datum.
PERIOD OF RECORD: December 1965 to March 1975, February 1977 to current year. Records for 1965 to 1975 are unpublished and are available in files of New Jersey District Office.
EXTREMES FOR PERIOD OF RECORD: Highest water level, 48.37 ft. (14.743 m) below land-surface datum, May 28, 1966; lowest, 104.34 ft. (31.803 m) below land-surface datum, between June 8 and September 29, 1982.
EXTREMES FOR CURRENT YEAR: Highest water level, 102.05 ft. (31.105 m) below land-surface datum, between October 1 and December 8; lowest, 104.34 ft. (31.803 m) below land-surface datum, between June 8 and September 29.

WATER LEVEL, IN FEET BELOW LAND SURFACE DATUM,
WATER YEAR OCTOBER 1981 TO SEPTEMBER 1982

WATER-LEVEL EXTREMES			MEASURED WATER LEVEL	
PERIOD	HIGHEST WATER LEVEL	LOWEST WATER LEVEL	DATE	WATER LEVEL
	LEVEL	LEVEL		
1 Oct 1981 to 8 Dec 1981	102.05	102.94	8 Dec 1981	102.53
8 Dec 1981 to 9 Mar 1982	102.44	103.37	9 Mar 1982	103.37
9 Mar 1982 to 8 Jun 1982	103.10	103.48	8 Jun 1982	103.16
8 Jun 1982 to 29 Sept 1982	102.91	104.34	29 Sept 1982	104.34

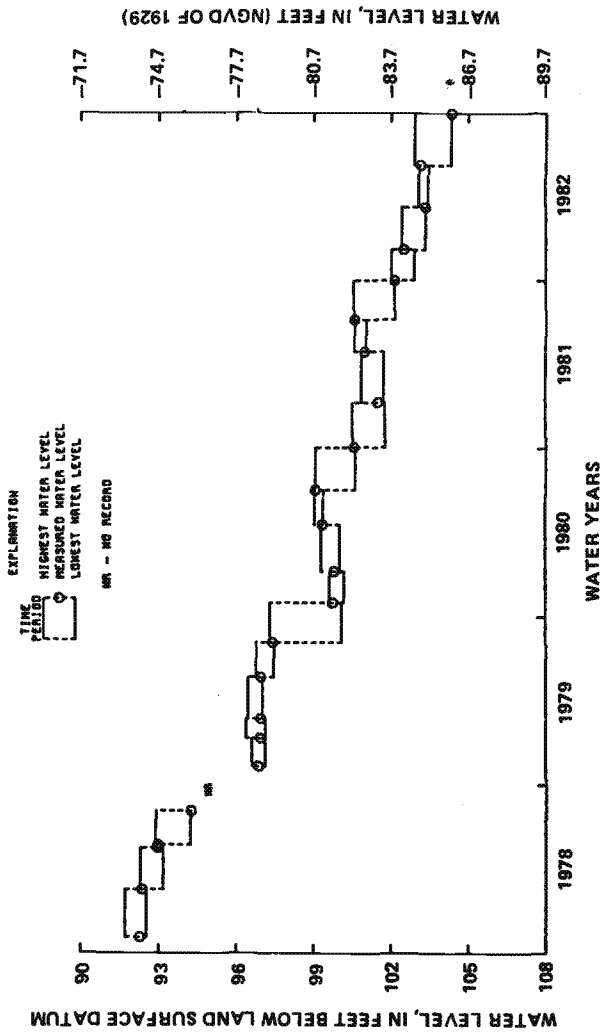


FIG. 3—Water level data for WRD Well No. 29—0334, Ocean County, NJ [4] (1 ft = 0.3048 m; 1 in. = 25.4 mm).

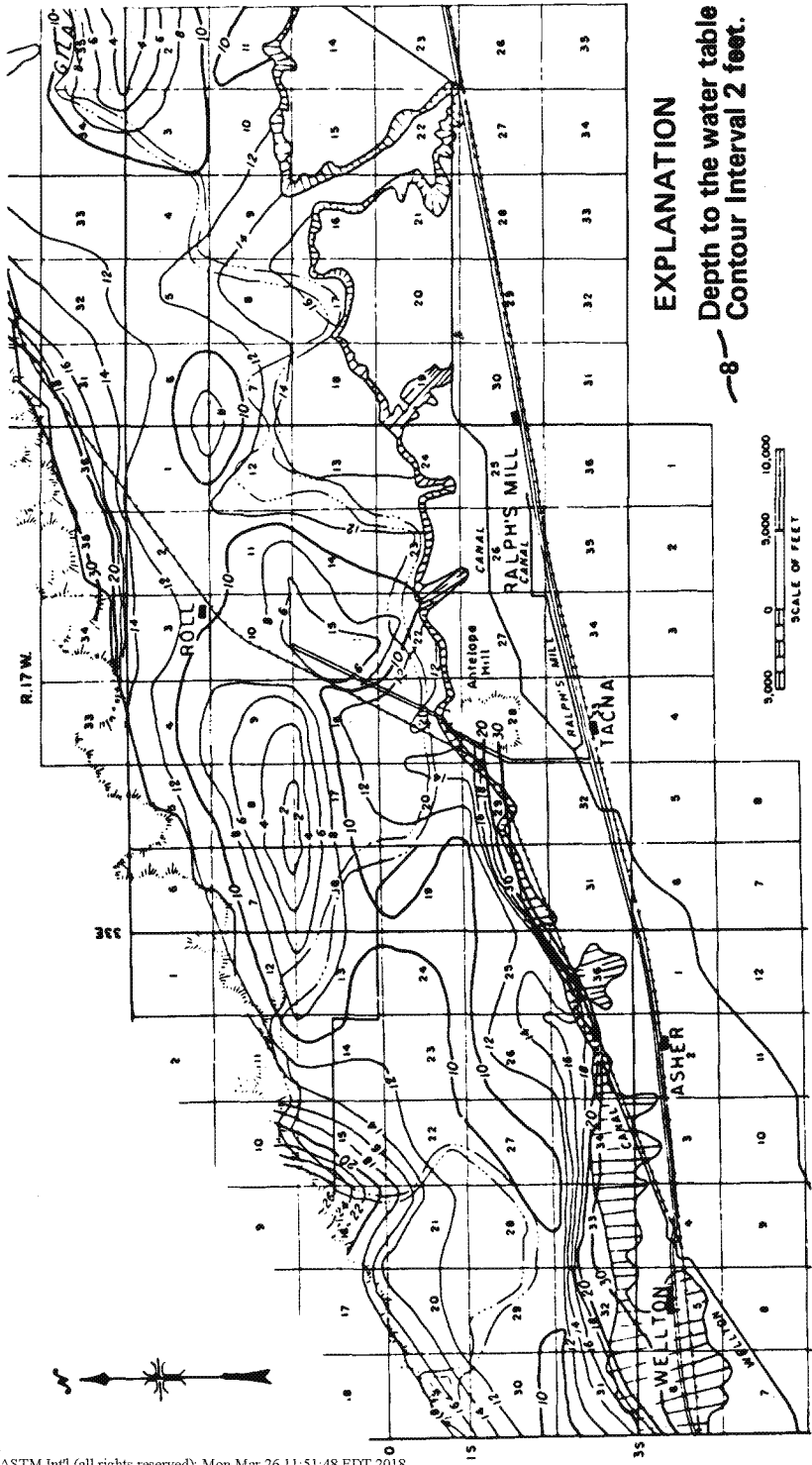


FIG. 4a—Depth-to-water-table contour map [5] (1 ft = 0.3048 m).

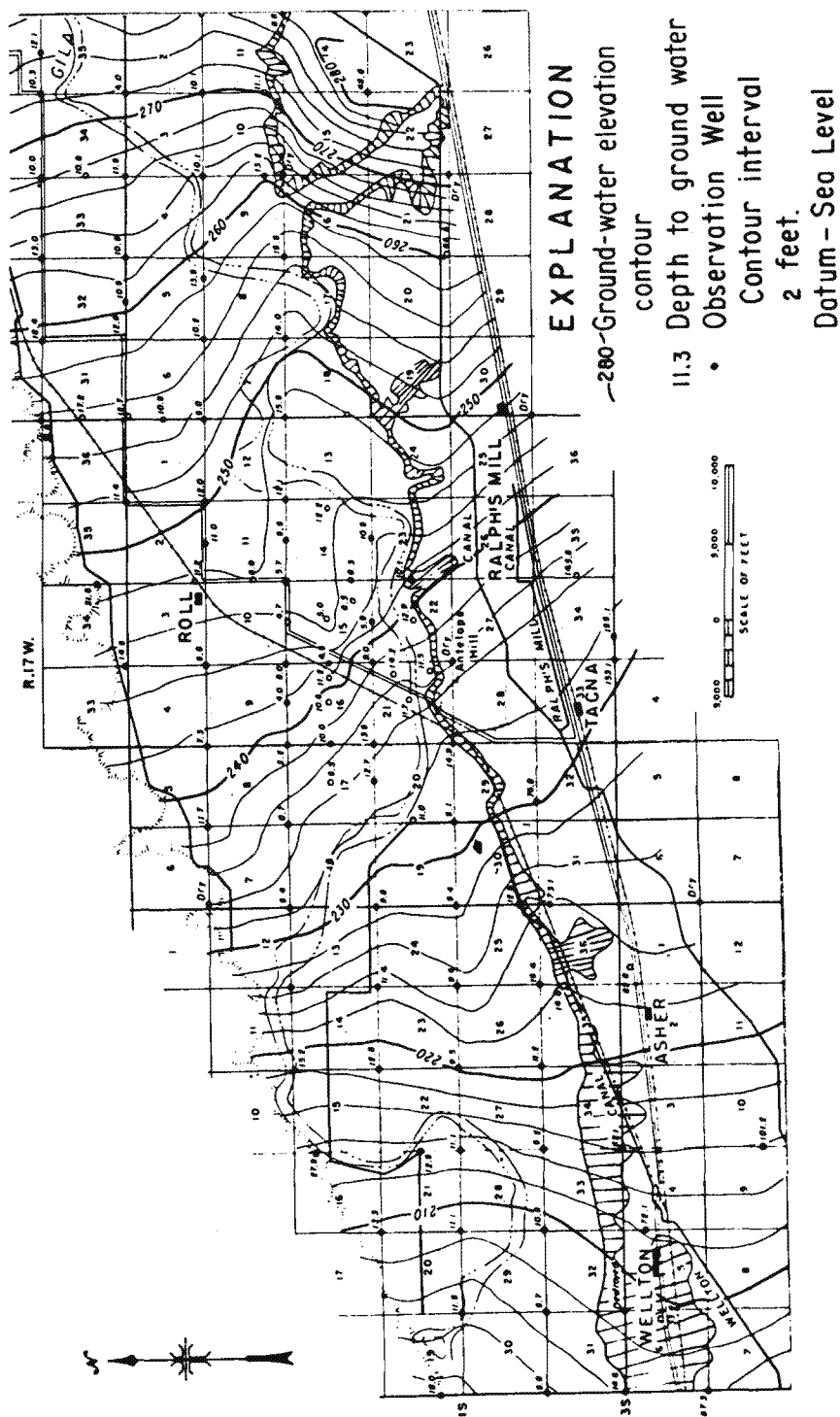


FIG. 4b—Water-table contour map [5] (1 ft = 0.3048 m).

with a water-level extremes recorder. This graph clearly shows that water levels continued to decline between 1978 and 1982. The quarterly measured water levels would depict a very different story if the extreme values were not shown. If one connects the quarterly values, it would appear that a considerable amount of recharge was taking place, occasionally superimposed on a steady decline. The extremes indicate less variation and fewer apparent upward trends.

Contour Plots, Flow Nets, and Sections

Contour plots in map view combine representative water levels from many monitoring points at a specified point in time. Synoptic water-level measurements are essential when an evaluation of ground-water flow is necessary. Figure 4a illustrates a depth-to-water-table map [5]. The same data have been converted to elevations above the sea level datum and presented in Fig. 4b. These two maps can be used for very different purposes. The depth-to-water-table map can be very important for geotechnical engineering and construction. It also gives the analyst an idea of where potential wetlands or other sensitive environments may be located. It can also be useful in planning and construction of deep foundations. The water-table elevation contour map is useful in evaluating ground-water flow. By measuring the spacing of the equipotential (contour) lines, the gradient between two points can be calculated. This information, when combined with the aquifer characteristics, allows the flow volume and velocity to be calculated. Lines constructed perpendicular to the equipotential lines indicate flow direction.

A plot with both equipotential lines and flow lines defines a flow net (Fig. 5). Figure 5 illustrates a ground-water mound caused by deep percolation (upper right corner). Water flows out radially from the mound. The distance between adjacent flow lines, W , increases with distance from the mound and with the flattening gradient. The size of the cells increases as the gradient decreases. The flow, V , through any cell equals the flow through any other cell. A flow net can be constructed in plan (map) view or in sectional (vertical) view. Flow nets are often used in engineering studies, especially when there is a contrast in material characteristics. They are also most useful when the horizontal or vertical gradients are steep and velocity profiles are of concern. Commonly, they are used to define seepage pathways either under or through an engineered structure, such as a dam. Flow nets can provide simplistic presentation of recharge and discharge relationships in a system.

Hydrogeologic profiles are constructed by combining geologic data from borings, test pits, and outcrops with water level data from observation points. Figure 6 shows a representative profile illustrating the relationship between a canal recharging the soils and the seepage discharging to a river. The differences caused by variations in material characteristics can be determined from the calculated slope of the water table at different points along the flow path. Presentations of this type are commonly found in geotechnical engineering studies, particularly because of the importance of water content in the strength of soil materials.

Summary

In summary, there are many ways to present ground-water level data so that the information can be understood and used. Commonly used x - y plots (hydrographs) are invaluable in visually evaluating trends in relation to time or distance. Contour maps give a snapshot view of the water table in terms of space. Flow nets and sectional views illustrate relationships that have wide engineering or hydrogeologic application.

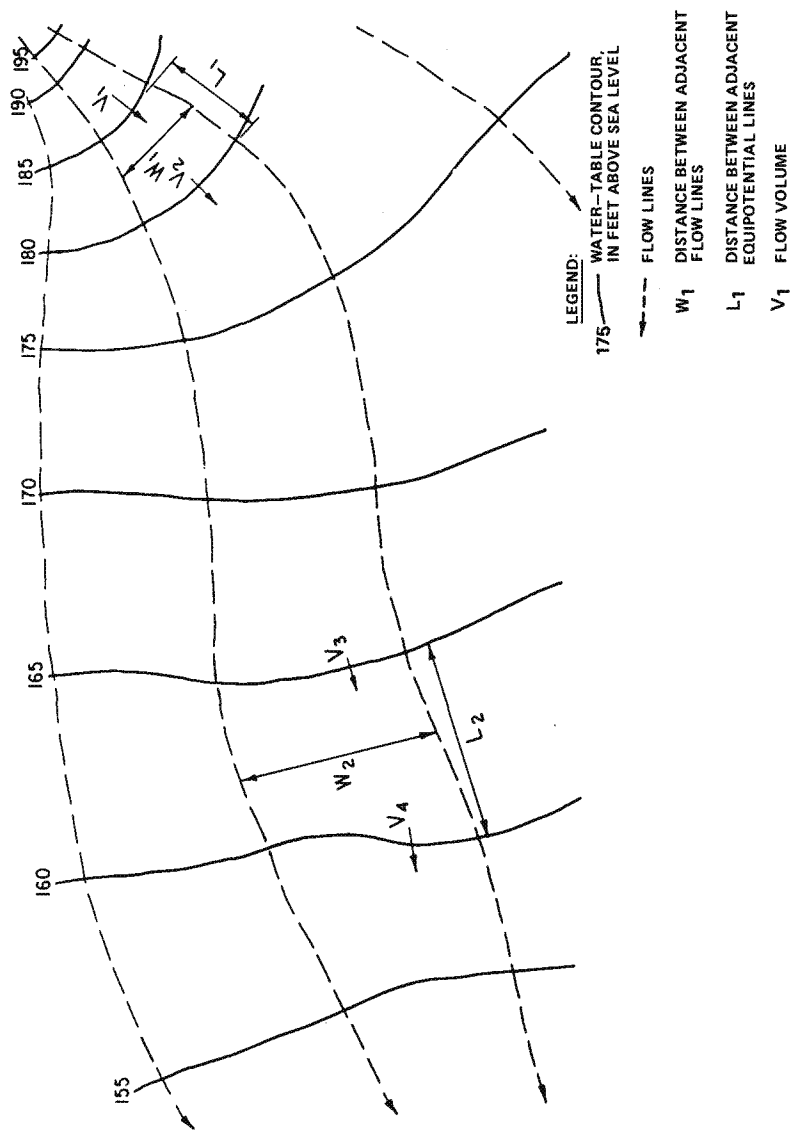


FIG. 5—Flow net analysis [5]. (The measurements are in feet (1 ft = 0.3048 m).

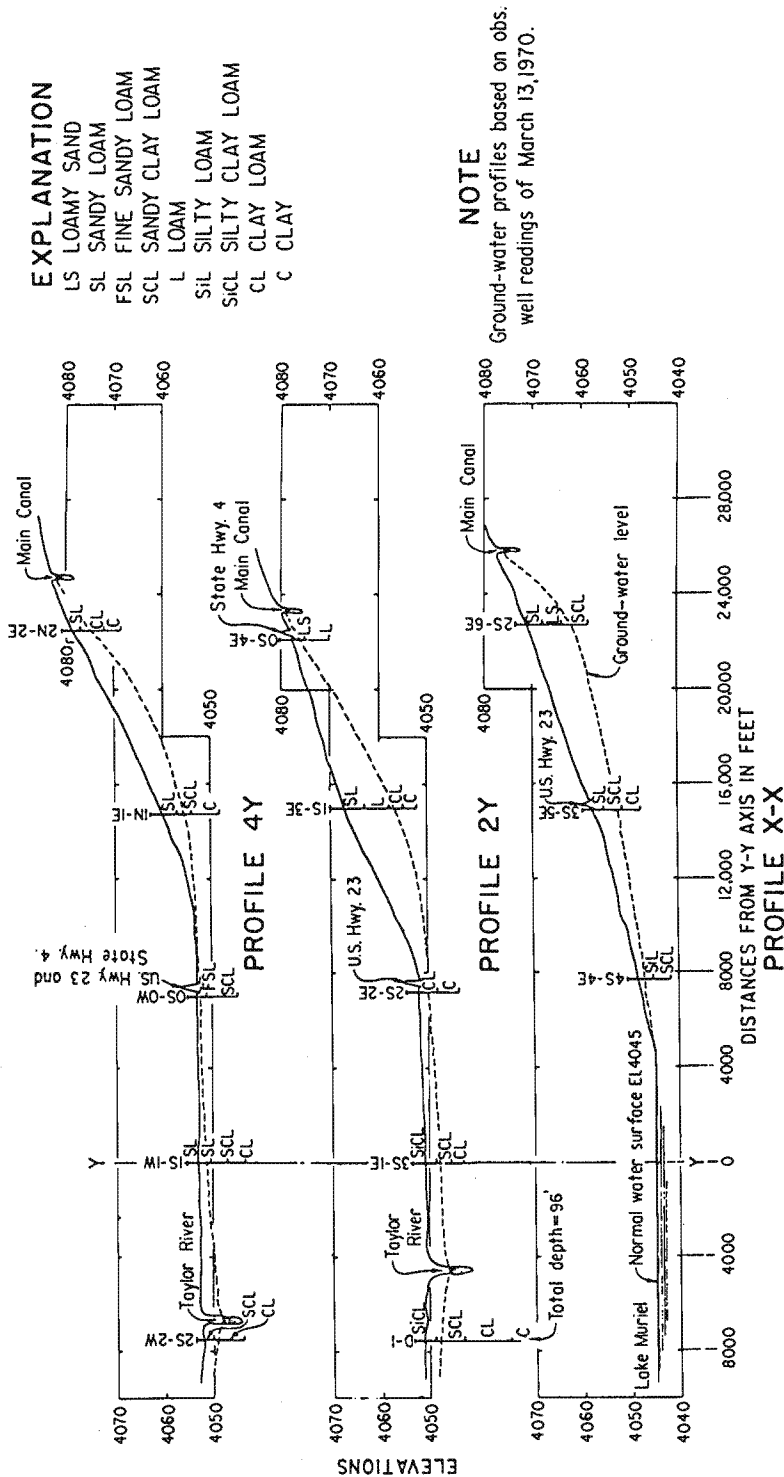


FIG. 6—Hydrogeologic profiles [5]. The elevations are the number of feet above sea level (1 ft = 0.3048 m).

References

- [1] Barry, B. A., *Engineering Measurements*, Wiley, New York, NY, 1964.
- [2] *ASTM Manual on the Presentation of Data and Control Chart Analysis, ASTM STP 15D*, American Society for Testing and Materials, Philadelphia, 1976.
- [3] Klein, M. and Kaser, P., "A Statistical Analysis of Ground-Water Levels in Twenty Selected Wells in Ohio," *Ohio Department of Natural Resources, Division of Water Technical Publication*, No. 5, 1963.
- [4] "Water Resources Data—New Jersey—Water Year 1982," USGS Water-Data Report NJ-82, U.S. Geological Survey, Denver, CO, 1982.
- [5] *Ground Water Manual*, Bureau of Reclamation, U.S. Department of the Interior, U.S. Government Printing Office, Washington, D.C.

Monitoring Well Purging and Ground-Water Sampling

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A Comparison of Two Multiple-Level Ground-Water Monitoring Systems

REFERENCE: Ridgway, W. R. and Larssen, D., "A Comparison of Two Multiple-Level Ground-Water Monitoring Systems," *Ground Water and Vadose Zone Monitoring, ASTM STP 1053*, D. M. Nielsen and A. I. Johnson, Eds., American Society for Testing and Materials, Philadelphia, 1990, pp. 213-237.

ABSTRACT: In the context of the Canadian Nuclear Fuel Waste Management Program, Atomic Energy of Canada Ltd. (AECL) is conducting research into the underground disposal of nuclear fuel waste deep in stable plutonic rock. As part of this research, AECL has developed and tested many geotechnical instrumentation systems. Two multiple-level ground-water monitoring systems have been used to evaluate hydrogeological conditions in fractured rock and overburden materials: the Waterloo system and the Westbay system. Both instruments are designed to isolate several sections of a single borehole for ground-water measurements. The instruments were tested in adjacent boreholes drilled to about 60-m depth in granitic rock at the AECL's Atikokan research area in Ontario, Canada. Various aspects of these instrument systems, including the design, testing and sampling methods, and ground-water measurement data are compared. The information contained in this paper may assist future users to determine if these systems, or other systems, will be effective for their applications.

KEY WORDS: Waterloo system, Westbay system, ground water, multiple-level monitoring systems, instrumentation design, installation activities, pressure measurements, fluid sampling, hydraulic conductivity measurements

Atomic Energy of Canada Ltd. (AECL) has been conducting hydrogeological research as part of the Canadian Nuclear Fuel Waste Management Program [1]. The objective of the program is to assess the concept of deep underground disposal of nuclear fuel waste in stable plutonic rock. Since instrumentation will be required to monitor ground-water conditions in the rock surrounding the underground disposal vault, AECL geoscientists have developed and evaluated many different instrumentation systems.

In 1985, AECL and Westbay Instruments Ltd. conducted a joint project to compare the Westbay and Waterloo multiple-level ground-water monitoring systems. The project was carried out at a study area at AECL's research area, located on the Eye-Dashwa granitic pluton, about 30 km north of Atikokan, Ontario, Canada (Fig. 1). The study area is situated on the Eye-Dashwa lineament (Figs. 2a and 2b), a northeast-trending regional topographic depression passing through the bottom of the Eye and Dashwa Lakes.

In this paper, the authors compare and evaluate various aspects of these two monitoring systems, including the instrument design, testing and sampling methods, and ground-water measurement data. We also review quality assurance procedures and costs. Although

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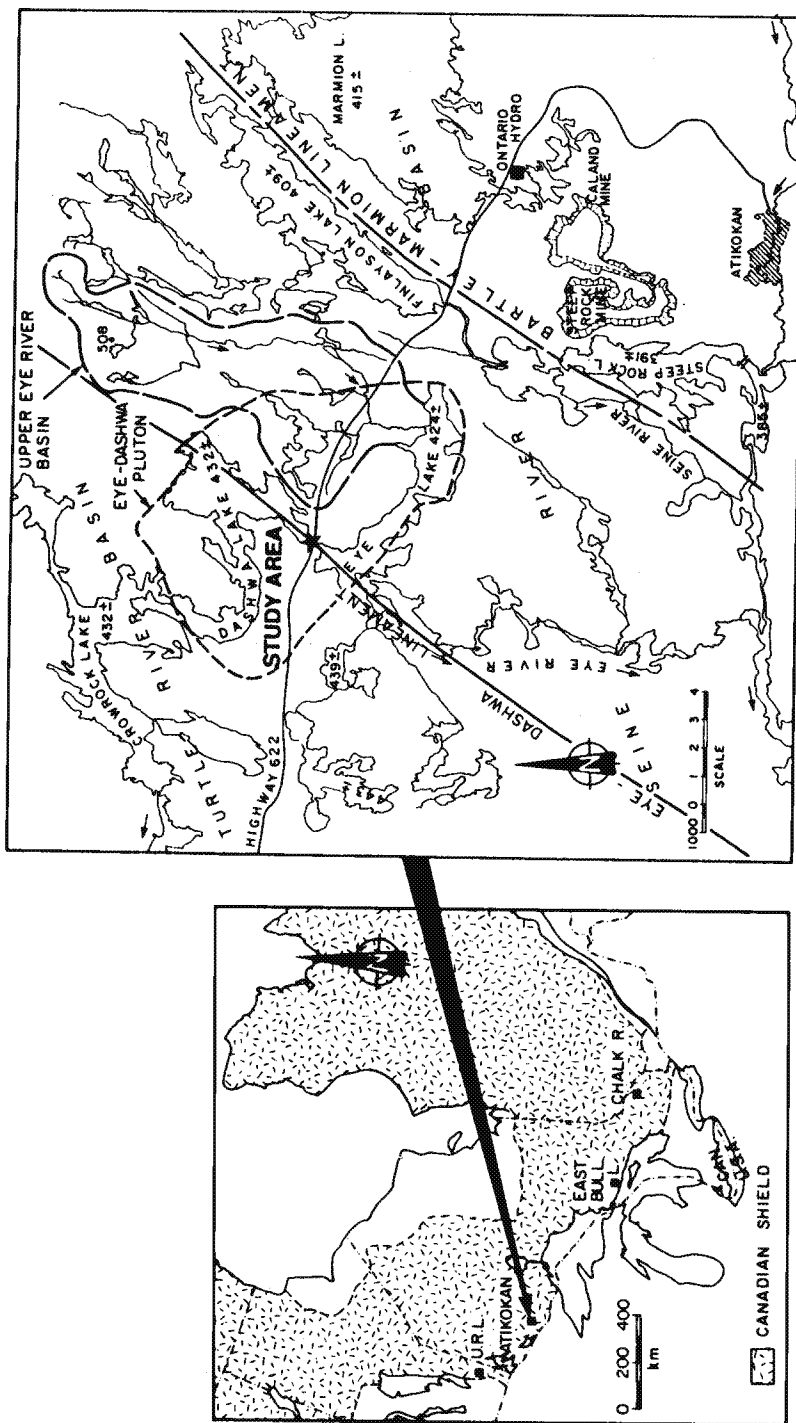


FIG. 1—Location of the AECL's Atikokan research area.

design changes and improvements have recently been made to both instrumentation systems, the basic operating principles and components are similar to those described in this paper.

Geology and Drilling

Two adjacent boreholes, ML-11 and ML-16, were placed 8 m apart on the portage separating the Eye and Dashwa Lakes. Figure 2*b* shows the local topography, the Eye-Dashwa lineament, the lake levels, and the ground-water springs, and Fig. 2*a* shows a cross section with the location of the two boreholes.

The two comparison boreholes, ML-11 and ML-16, were drilled vertically to a depth of 56 and 59 m, respectively, using a track-mounted Longyear 38 drill rig with NQ-size (76-mm-diameter) diamond coring equipment. Clear surface water (that is, containing no drilling mud or drilling fluid) was used to minimize contamination of the formation ground water. During drilling, the drill water was labelled with rhodamine dye at selected depth intervals, to aid in subsequent purging activities. The drill core was logged, and lithology and discontinuity data were collected [2].

The bedrock in ML-11 and ML-16 consists of fractured, partly weathered to unweathered granite. The fracture discontinuity index ranges from 0 to more than 20 per metre and the rock quality designation (RQD) values [3] range from about 50 to 100%. The fracturing is discussed in the sections on ground-water chemistry and hydraulic conductivity testing, further on in this paper. Overburden materials consist of sandy and silty glacial outwash deposits, 5.9 m deep at ML-11, and 9.0 m deep at ML-16. The Waterloo and Westbay instruments installed in these boreholes had access ports located in both bedrock and overburden materials.

Instrument Components and Configuration

Figure 3 is a schematic diagram showing components of the Waterloo and Westbay instrumentation systems (based on figures presented by Cherry and Johnson [4] and by Black et al. [5]).

Waterloo System

The ground-water monitoring system of the University of Waterloo [4], Waterloo, Ontario, Canada, was installed in borehole ML-11. The installed system was a prototype, different from the current version of the Waterloo system. (Shortly after this installation was carried out, the manufacturing rights for the Waterloo instrument were obtained by Solinst Canada Ltd., and an updated commercial version became available in mid-1985.) The instrument consisted of a bundle of open, flexible polyethylene access tubes. Each tube, 10 mm in inside diameter (ID) and 13 mm in outside diameter (OD), was connected to an individual monitoring zone. The tubes were contained within a 51-mm-ID Schedule 40 polyvinyl chloride (PVC) casing.

The monitoring zones of the Waterloo instrument were isolated by self-inflating chemical packers clamped on the outside of the casing. The packers consisted of an expandable Dowell³ sealant sleeve covered with a waterproof rubber membrane. When water was

³ Dowell sealant was obtained from the Dow Chemical Co.

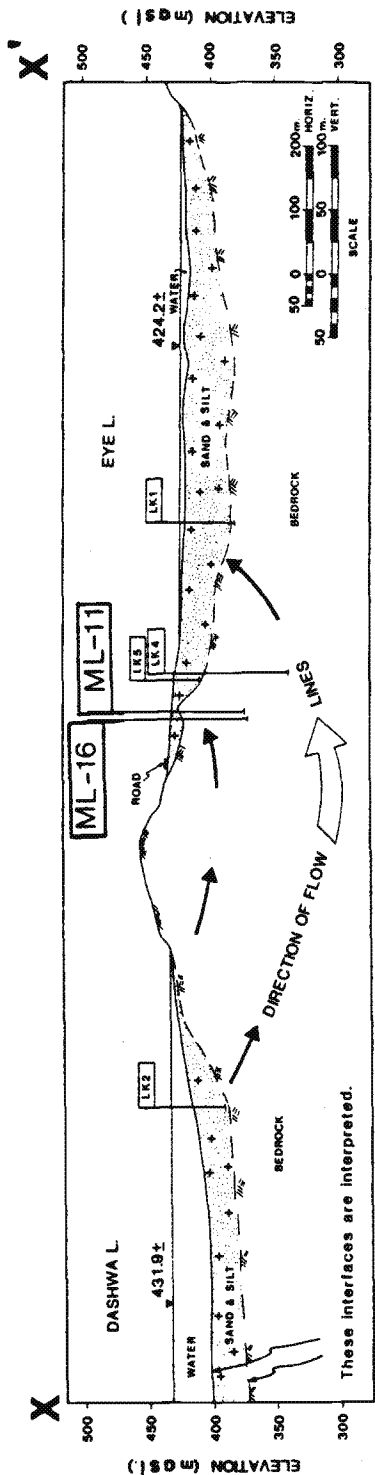


FIG. 2a—Cross section of the study area.

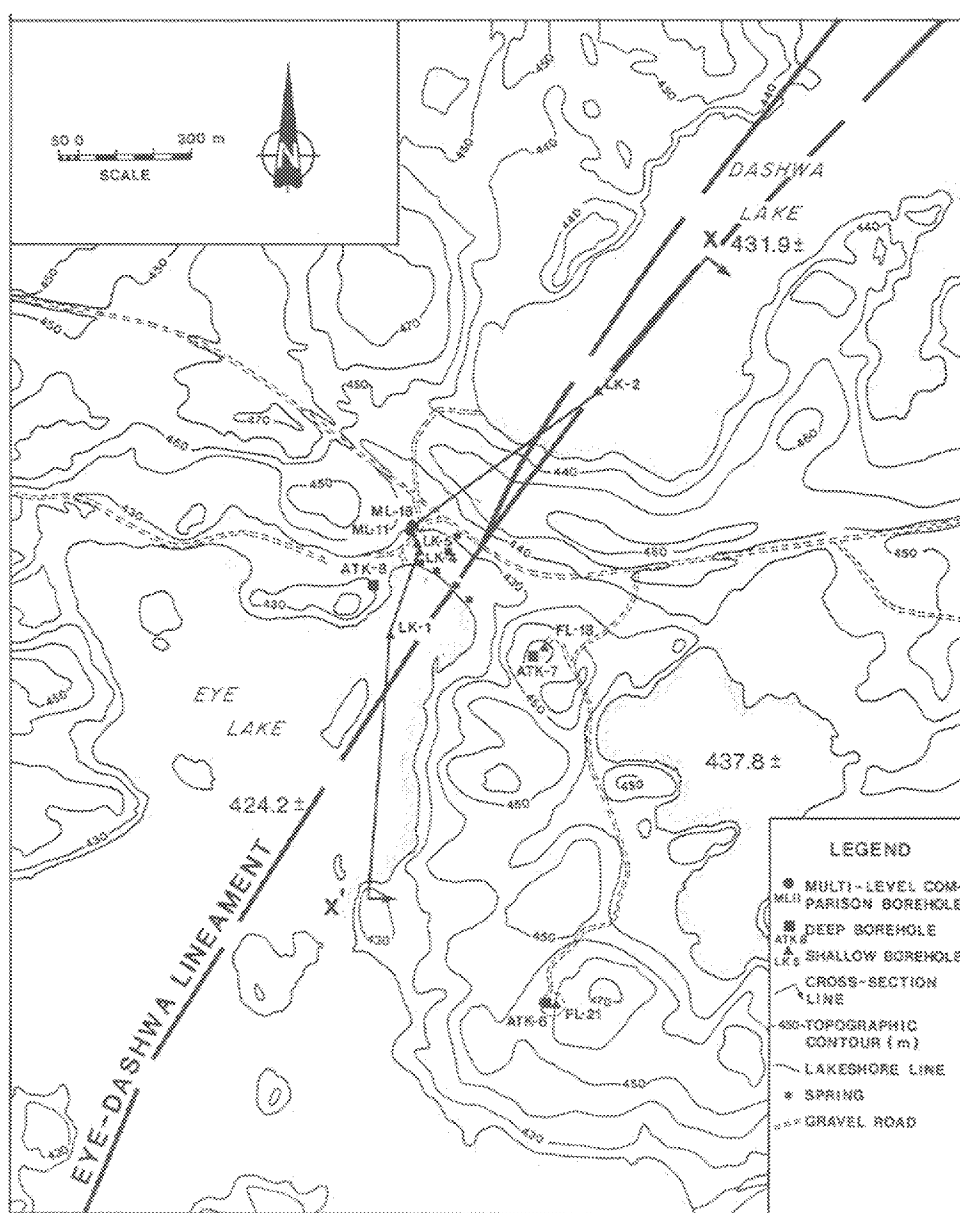


FIG. 2b—Plan view of the study area.

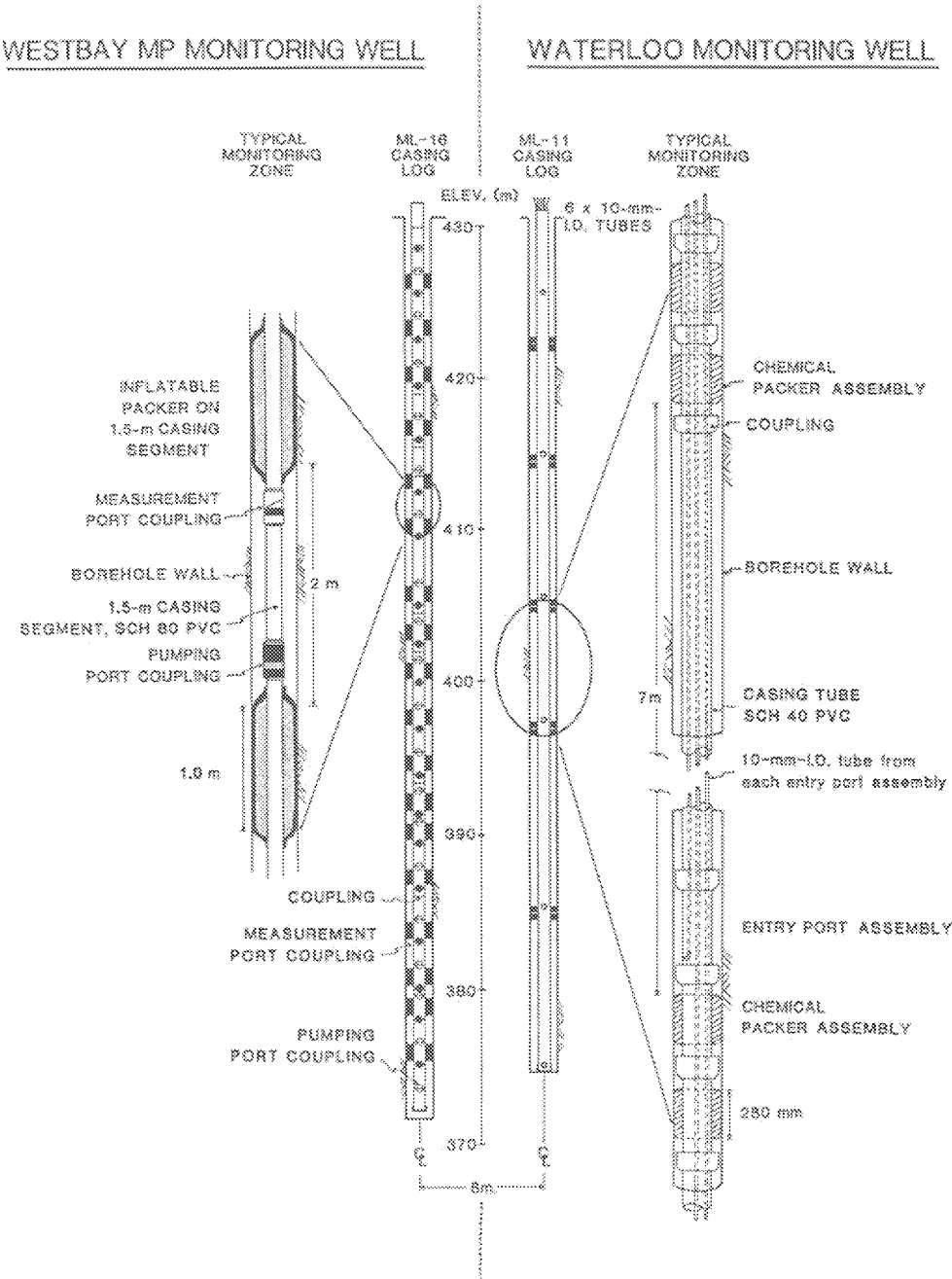


FIG. 3—The Westbay and Waterloo (ML-16 and ML-11) comparison boreholes, showing the two instrumentation systems.

poured into the interior of the PVC casing the packers expanded. The water came in contact with the Dowell sealant sleeves through holes in the wall of the casing (not shown in Fig. 3). This water contact caused expansion of the sleeves to more than twice their original volume, sealing off an NQ-size (76-mm) borehole. The packers were fully expanded after about 24 h. The installed instrument consisted of ten packer assemblies (Fig. 3) arranged in five pairs, to isolate six monitoring zones. Each pair of packers had a total length of about 800 mm, and the length of the individual monitoring zones ranged from 7 to 11 m.

The polyethylene tubes provided access for monitoring and sampling ground water from the designated zones. Each access tube was joined to an entry port and extended from the port to the ground surface. The entry port was an open elbow (10-mm OD) of PVC pipe that penetrated through the casing.

The PVC casing segments were threadless and the string was joined together with glued couplings. The installed instrument string had (1) a Schedule 40 plug glued on the bottom end, (2) six entry port/access tube assemblies, (3) five packer assemblies, and (4) several PVC pipe segments. In the updated Solinst version of the Waterloo instrument, double O-ring seals replace glued joints.

Westbay System

Borehole ML-16 was completed with a Westbay ground-water monitoring system (Fig. 3), which was designed and manufactured by Westbay Instruments Ltd. The first commercial version of the Westbay instrument was introduced to the geotechnical-hydrological field in 1978. The current Westbay system components and probes are described by Black et al. [5]. The Westbay installation consisted of a single, closed access tube made of 37-mm-ID Schedule 80 PVC pipe segments with machined ends for coupling attachment. The three different coupling types used in this installation were each formed of acrylonitrile-butadiene-styrene (ABS) plastic with Viton O-ring seals. A shear wire was threaded through the couplings to join the casing segments.

The monitoring zones of the Westbay instrument were isolated by hydraulically inflated packers attached to the outside of the casing segments. The packers consisted of a urethane gland with stainless steel clamps at either end of a PVC casing segment. These were inflated individually with a packer inflation tool, which was lowered into the instrument casing. The tool was seated on a one-way valve, which allowed movement of fluid through the wall of the casing and into the interior of the packer gland. The packer inflation was controlled from the surface by measuring the fluid injection pressure and volume at selected time intervals. The packer sealing pressure was about 690 kPa (100 psi), and each packer had a sealing length of about 1 m.

The instrument was installed (Fig. 3) with two different types of valved couplings for access to each monitoring zone. Fluid samples and fluid pressures were taken through the measurement port couplings using a probe, which was lowered inside the instrument casing. Purging, permeability testing, and large-volume sampling were carried out through the pumping port couplings. A mechanical wire-line tool was used to open and close the valves on the pumping port couplings. When a single pumping port was opened, only one monitoring zone was connected to the casing and the casing behaved like a single conventional standpipe piezometer. The modular design of the couplings and casing segments allowed installation of 19 monitoring zones into borehole ML-16. These zones ranged from about 1.0 to 4.0 m in length and were isolated by 18 packers.

Installation Methods

The monitoring zones and annulus seals for the two boreholes were selected by reviewing the geological drill logs. The instruments were installed during winter, and hence a heated drill shack was required during drilling and instrumentation installation. After completion of the borehole drilling, each hole was flushed for several hours with surface water to remove any drill cuttings. A whole day was set aside for each installation. Considering the winter conditions, which were particularly adverse during the Waterloo system installation, both installations went smoothly and successfully.

Waterloo

For the Waterloo instrument, a casing log was prepared the night before the installation. The instrument casing and tubing were cut to length, and the other parts were assembled and labelled. On the installation day, the sections of PVC casing were laid out in sequence next to the borehole. The couplings, rolls of polyethylene tubes, and packer assemblies were retained in the heated drill shack. The installation was carried out on 27 Feb. 1985, with the outside air temperature at -30°C .

The installation began with attaching and clamping the longest polyethylene access tube to the lowest entry port assembly. Then, the tubing was threaded through a 3-m section of casing, and the port assembly and casing were joined with a coupling. The coupling and casing were heated, glued together, and placed into the borehole. Two more segments of PVC casing, a packer assembly, and an entry port assembly were threaded, coupled, and carefully glued together. This was followed by attaching and clamping the next longest polyethylene tube to the next entry port assembly. The threading and coupling of components was continued until the installation was completed.

The instrument string was watertight and buoyant when placed below the water level in the open borehole. During the installation, surface water was poured into the casing to overcome the buoyancy effect. The Waterloo installation was completed in 4 h with four people participating. Next, the casing was filled with water, causing the packers to expand and seal the borehole annulus. The next day, tests were carried out to check the depth of the access ports and the borehole annulus seals.

Westbay

For the Westbay instrument, a casing log describing the position of each of the instrument components was also prepared the night before the installation. The component supply was checked to ensure that all the required parts were available. Each coupling of the instrument string had a serial number, which was recorded. On the installation day, the couplings were attached to the casing segments and these were laid out in correct sequence next to the borehole. The instrument components were coupled together and the string was placed into the borehole under the shelter of a drill shack. During installation of the instrument string, surface water was periodically poured into the casing since the casing was watertight and buoyant.

The Westbay instrument string was installed on 25 March 1985, in about 4 h with three people participating. The outside air temperature was $+2^{\circ}\text{C}$. The following day, an additional 10 h was needed to inflate the 18 hydraulic packers sequentially using a packer inflation tool, beginning at the lowest packer. The volume of water injected into the packers and the packer inflation pressures were recorded for quality control purposes. During the inflation period, the instrumentation couplings were also located with the inflation tool.

When the tool was pulled upward from the bottom of the instrument string, it seated itself on the helical shoulders located on the inside of each coupling.

Fluid-Pressure Measurements

Waterloo

The method for fluid-pressure measurement in the Waterloo instrument was the same as that used in an open standpipe piezometer. A small-diameter (5-mm) electric water-level tape was used to measure the water level in each polyethylene access tube, and these measurements were compared with a known benchmark elevation. The average time required to measure water levels in the six monitoring zones of ML-11 was about 15 min, or 2 min per measurement. More time was needed to test whether the annular seals were performing properly.

During winter, the water level in the interior of the casing was drawn down below the frost line to prevent the instrument from freezing. However, the instrument had to be thawed and a heated enclosure was required for winter monitoring since the water levels in the access tubes were above the ground surface.

Westbay

The method for fluid pressure measurements in the Westbay instrument involved lowering a probe, containing an electric pressure transducer, inside the casing to a measurement port coupling. Initially, the probe was lowered to the bottom of the instrument string and was pulled upward until it was seated on the desired measurement port coupling. Then, it was activated to complete a hydraulic connection to the monitoring zone. During activation, a seal was made isolating the pressure transducer from the fluid inside the instrument casing. Once the transducer had stabilized, the pressure was displayed on the transducer control unit at the ground surface.

The pressure at each monitoring zone was measured in sequence, beginning at the bottom of the casing. The piezometric pressure was calculated based on the difference between the readout measurement and the reference pressure and elevation of the interior fluid column. The average time required to measure the pressures at all 19 monitoring zones was 60 min, or about 3 min per measurement.

During winter, pressure measurements were carried out without the instrument having to be thawed. The water in the access pipe was isolated from the monitoring zones and was maintained below the depth of freezing.

Fluid Pressure Data

The piezometric profile from the Waterloo instrument (Fig. 4) indicates a component of upward flow, from 423 m to the water table at 428 m, stable artesian conditions between 423 m and 406 m, a downward component between 406 and 405 m, stable artesian conditions from 405 m to 385 m, and a slight upward gradient in the lowest interval (all measurements are in metres above mean sea level).

The piezometric profile of the Westbay instrument (Fig. 4) is more detailed since more zones were installed. For the upper part of the hole from about 430 to 412 m, the readings are in agreement with those from the Waterloo instrument. Below 412 m, the readings are slightly different, and from 412 to about 405 m there are stable artesian to weakly downward gradients. This is underlain by a strong downward gradient to about 400 m. From

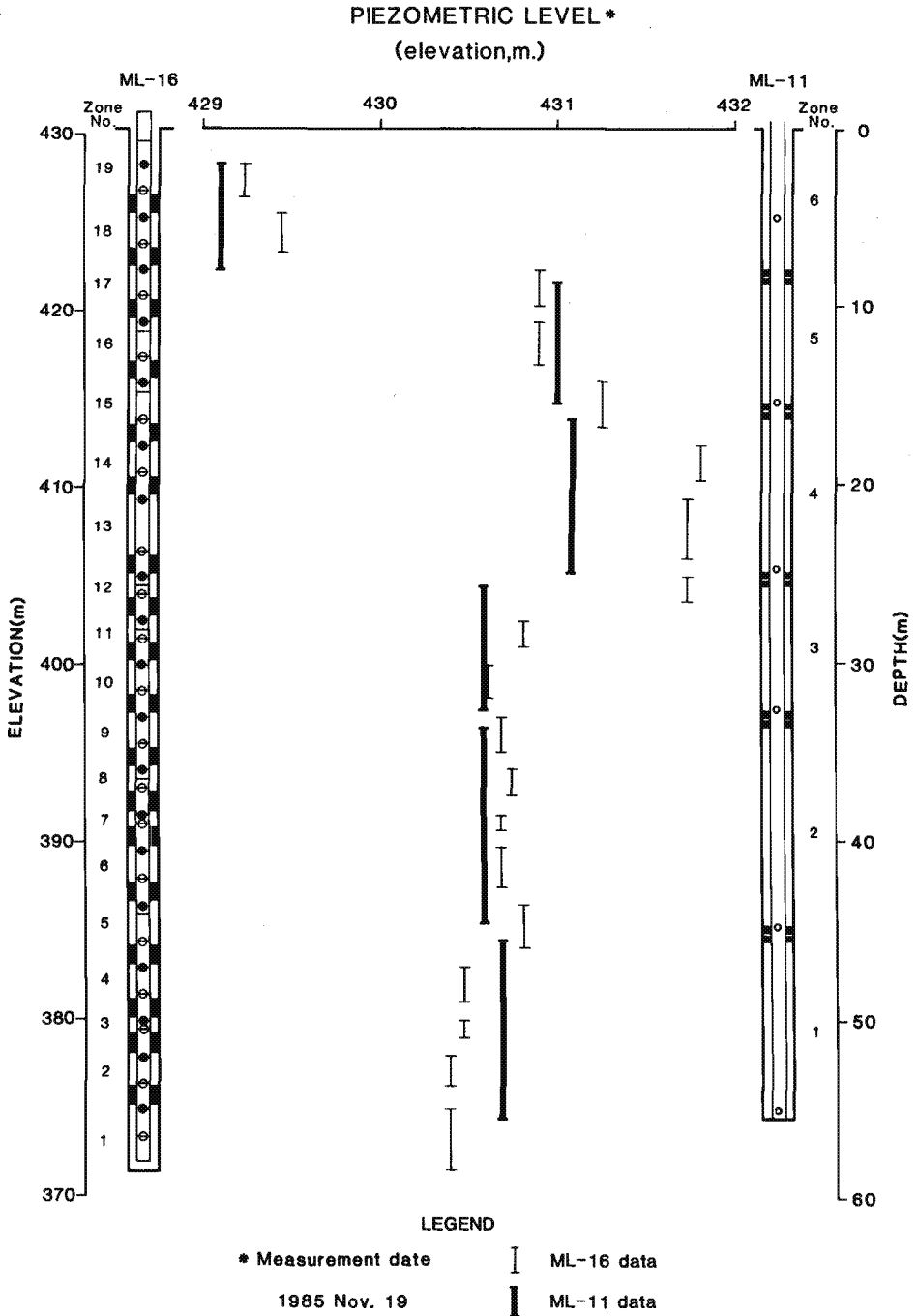


FIG. 4—Comparison of the piezometric profiles for ML-16 and ML-11.

about 400 to 384 m, there are stable artesian conditions, and below 384 m the gradients are stable to weakly downward.

The piezometric data from the Westbay instrument showed a wider range of measurements than that from the Waterloo instrument. The range is probably in response to water pressures from discrete fractures, which can be measured with short monitoring intervals. Generally, short intervals minimize interzone flow and therefore reduce any effect of piezometric-level averaging. Averaging may have occurred in ML-11 as a result of the long monitoring zones. There are inflections in the ML-16 profile at depths of 18, 27, 30, and 47 m, and these are consistent with the packer seals in ML-11 at depths of 16, 26, 34, and 46 m. The monitoring zones in ML-11 may have caused short-circuit effects to nearby ML-16. Alternatively, the differences in piezometric readings may have been caused by differences in fracture hydrology between the two boreholes.

Fluid Sampling Methods

When possible, a similar sampling procedure was used for both instrument systems. Development pumping and purging were carried out to remove drilling fluids from each monitoring zone. The absence of rhodamine dye in the purge discharge water from all monitoring zones in both instrument systems indicated that the drilling fluid was no longer present. After development and purging activities had been completed, water samples were taken from most zones of both wells as follows:

ML-11: Large-volume samples were collected from 6 of 6 zones.

ML-16:

1. Large-volume samples were collected from 6 of 19 zones (Nos. 2, 6, 10, 13, 16, and 17). These zones were selected since they closely correspond to the 6 zones in ML-11.
2. Small-volume samples were collected from 12 other zones.
3. Zone No. 9 was not sampled.

Waterloo

The standing-fluid levels of the monitoring zones in the Waterloo instrument were above the ground surface, and thus, purging could be done using a peristaltic suction lift pump. A line from the pump was placed into the polyethylene access tube and used to purge water from the tube and the monitoring zone (Fig. 3).

Purging was carried out during August 1985 and the data from this activity are presented in Table 1. The purge volume⁴ for each monitoring zone ranged between 12 and 22 L, and about 0.2 to 1.3 times these volumes was pumped from the zones. The average pumping rate was about 69 mL/min. This rate was limited by the low transmissivity of the monitoring zone and the suction lift capability of the pump. Complete development of this monitoring instrument for removal of drill cuttings was not possible because of the low flow rates and a time constraint. This lack of development, however, was not considered to have hindered the instrument performance since the hole had been extensively flushed prior to installation of the instrument.

⁴ The purge volume was calculated as the sum of the water in the annulus between the borehole wall and casing, and the riser tube volume.

TABLE 1—Purging data for the two instruments.

Well No.	Monitoring Zone	Date	Purgeable Volume (L)	Pumping Rate (ml/min)	Pumping Time (min)	Volume Pumped* (L)	Purgeable Volumes Pumped (L)
ML-16 (Nov. 1985)	Number						
	1 to 12	19, 20, 21	145	1700	640	1090	7.5
	13 to 19	21	95	2000	360	720	7.6
	13 to 19	22	95	1000	420	420	4.4
Total			240		1470	2230	
ML-11 (Aug. 1985)	1	6	21.5	228	120	27.4	1.27
	2	6, 7	22.4	42	150	6.3	0.28
	3	7, 8	14.7	33	90	3.0	0.20
	4**	8, 9, 12	16.6	15	155	2.3	0.14
	5	8, 9	14.6	33	95	3.1	0.21
	6	9	12.2	67	60	4.0	0.33
Total			102		670	46.1	

* Prior to sampling.

** Pumped dry, insufficient recharge for sampling.

Westbay

The fluids were purged and the monitoring zones were developed in the Westbay instrument (Table 1) by using an electric centrifugal pump. The pump was attached to a 19-mm PVC intake line placed at the bottom of the instrument casing. The water in the monitoring zones was accessed by opening the valved pumping port couplings (Fig. 3).

The instrument monitoring zones were purged once and in two groups, based on a review of fluid pressure readings. The first group of monitoring zones, Nos. 1 through 12, had similar fluid pressures⁵ in each of the monitoring zones. Before these valved couplings were opened, the water level inside the instrument casing had been drawn down to minimize any fluid mixing between the monitoring zones. After completion of purging of the first group of zones, the valves of these pumping port couplings were closed. The second group of monitoring zones, Nos. 13 through 19, were purged using the same method.

The purging data are presented in Table 1. The purge volume of the lower group of zones was 145 L, while that of the upper group was 95 L. During 19–21 Nov. 1985, an amount 7.5 times the purge volume was pumped from each group of zones. The upper group of zones was purged for a second time on 22 November, when an additional 4.4 times the purge volume was removed. With these activities, it is assumed that all monitoring zones, even those with low transmissivity, had had at least one purge volume of fluid removed.

A surge-block technique was used for development and purging of the Westbay instrument, to achieve sediment-free water discharge. This plunging technique caused water to move into and out of a pumping zone and was used to remove particulate matter. Purging of the instrument was carried out just after the installation of the instrument, since the monitoring zones and casing were hydraulically isolated from each other. Successive sampling did not have to be preceded by purging, but further purging could have been done by repeating the described process.

Sample Collection

Water samples were collected from all six monitoring zones in the Waterloo instrument. Water sampling was carried out using a peristaltic pump and with the same method as was described for purging. For each zone, the procedure involved pumping until steady-state water chemistry was achieved. When this occurred, water samples were taken from the pump discharge line. After completion of sampling, the pump lines were rinsed with deionized water.

Water samples were collected from 18 of the 19 monitoring zones in the Westbay instrument. Sampling was done with a stainless steel bottle connected to a wire-line-operated sampling probe. For each zone, the procedure involved obtaining a volume of water (80 to 250 mL) to rinse the collection apparatus. Then, the probe was repeatedly lowered to the same zone until the required volume of water was collected. After collection of a sample from a zone, the sampling apparatus was cleaned with a rinse of dilute nitric acid followed by a rinse of deionized water.

Sampling data for ML-11 and ML-16, based on sampling activities carried out in November 1985, are summarized in Table 2. The collection rate for ML-11 represents the average water discharge of the peristaltic pump calculated when the formation was yielding water. The time required to obtain a sample was greater than the collection rate indicates because of the intermittent pumping time. The collection rate for ML-16 is based on the

⁵ There was, however, about 1 m of hydraulic head difference between Zones 1 through 11 and Zone 12.

TABLE 2—*Sampling summary for comparison of the instruments.*

	Westbay (ML-16)	Waterloo (ML-11)
Sample collection rate	29 mL/min	37 mL/min
Time required per sample	49 min	167* min
Total volume of sample collected	14865 mL	5700 mL
Total sampling time	875 min	1084 min

* with incomplete purging.

time needed to retrieve samples from the measurement port with the sampling apparatus. The time required for each sample included three to four trips to the port and the preparatory activities.

Ground-water Chemistry

Ground-water samples were analyzed in the field using AECL’s mobile laboratory. Field measurements included the pH, temperature, conductivity, and chloride and bicarbonate concentrations. Some of the water samples were acidified and sent to AECL laboratories for detailed analyses.

The chemical results compared are the pH and electrical conductance (Fig. 5) and the total dissolved solids (TDS) and tritium values (Fig. 6). The trends for various chemical parameters are similar for both instruments. These values are typical for shallow ground water in the study area and comparable to a summary of chemical results contained in a report by Underwood McLellan Ltd. [2].

The pH values, which generally increase with hole depths, range from 6.1 to 7.8 and tend to be higher in less fractured rock. At some depths, the values for ML-11 appear to be an average of those values measured in several zones of ML-16. The averaging is probably the result of the longer monitoring zones in ML-11. The TDS and conductivity values also increase with hole depths and the values for both holes correspond closely to each other. However, from about 400 to 380 m in ML-16, there is a decrease in conductivity and TDS. At the same depth interval in ML-11, there is an increase in these values.

The tritium results from both wells are comparable and range from a low of 10 tritium units (TU) to a high of 59 TU. The low values at 400 m may be related to its being a lower hydraulic conductivity zone, since areas of lower hydraulic conductivity appear to have lower tritium content.

Hydraulic Conductivity Testing

The hydraulic conductivity for each monitoring zone of the two instruments was tested after completion of the purging and development activities. Hydraulic conductivity measurements were performed using a variable-head slug test [6].

In the Waterloo installation, either a rising-head or a falling-head hydraulic conductivity test could have been performed. A rising-head test technique was used because the static

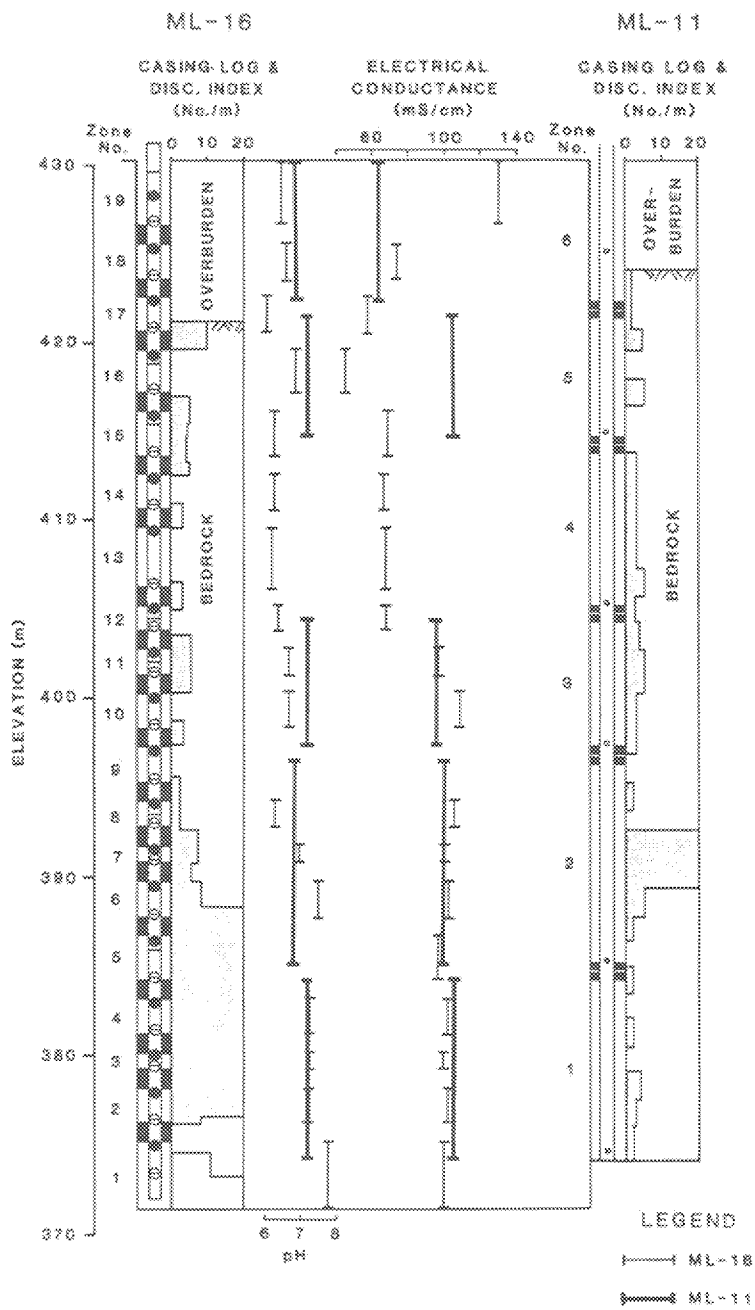


FIG. 5—Geochemical profile showing the pH and electrical conductance of ground waters from ML-11 and ML-16.

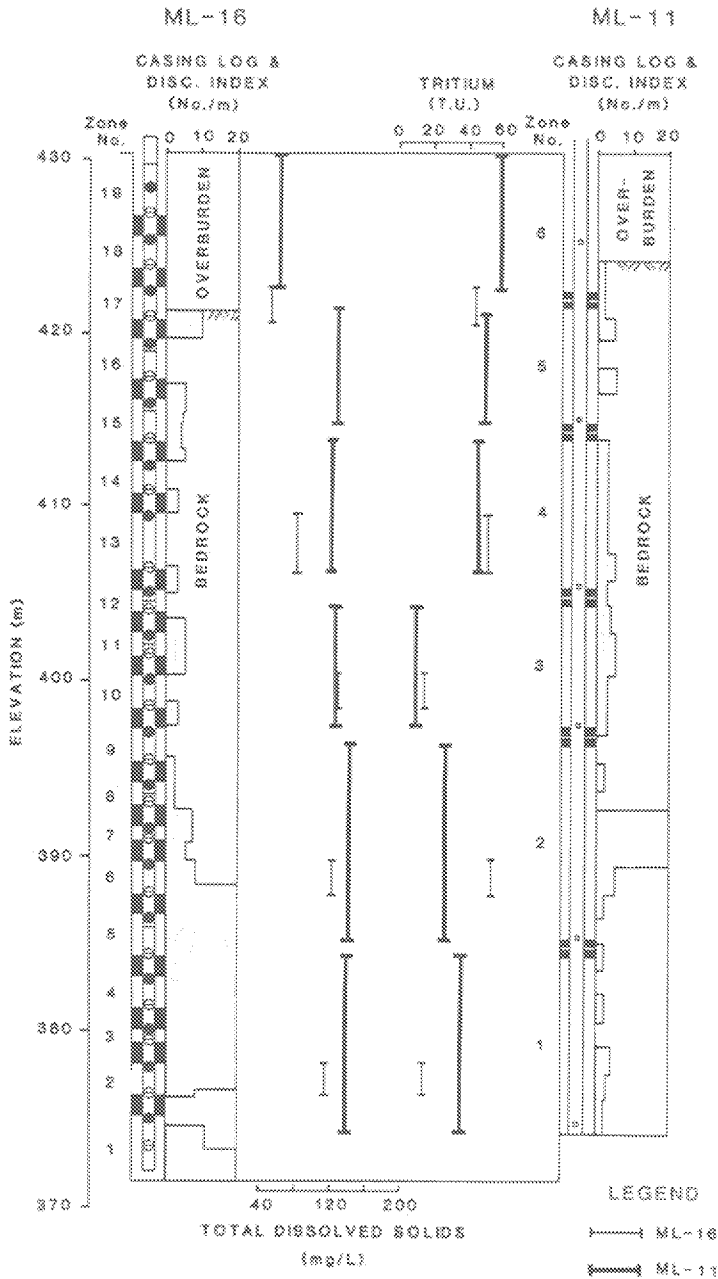


FIG. 6—Geochemical profile showing the total dissolved solids (TDS) and tritium in ground waters from ML-16 and ML-11.

water level was above the ground surface. The test was performed using a peristaltic pump and a measured length of suction line. The water level in the access tube was lowered to a depth of about 5 m below the static water level. The test was initiated when the suction tube was removed and substituted with a narrow-diameter water-level indicator. The recovery with time of the water level in the access tube was recorded.

In the Westbay installation, either a rising- or a falling-head hydraulic conductivity test could have been performed, as well, but a rising-head test was chosen so that the test methods in the two installations would be comparable. The test was performed using an electric centrifugal pump and a PVC intake line to lower the water in the instrument casing. Prior to each test, all of the valved couplings were closed, which made the instrument casing watertight. Then, the water level in the casing was pumped to a level of about 5 m below the piezometric level in the zone to be tested. The test was initiated by opening the desired pumping port using a wire-line tool. An electric water-level tape was used to measure the water-level recovery with time in the instrument casing.

Test Data

Hydraulic conductivity test results for ML-11 and ML-16 are shown in Fig. 7. The data from ML-16 correlate better with the degree of fracturing or discontinuity index than the data from ML-11. However, not all zones of intense fracturing in ML-16 have corresponding high hydraulic conductivity values. Hydraulic conductivity values (Fig. 7) from ML-11 are generally lower than those from ML-16. The ML-11 values range from 1×10^{-7} to 7×10^{-10} m/s and average about 3×10^{-8} m/s. The ML-16 values range from 9×10^{-5} to 7×10^{-9} m/s and average about 8×10^{-6} m/s.

Discussion

Table 3 summarizes the comparison between the two instrumentation systems.

Well Design

The downhole components of the Westbay and Waterloo monitoring instruments were mainly plastic (PVC and ABS), with some metal parts. The main differences between the systems were the monitoring zone access methods and the borehole annulus sealing method.

For the Westbay instrument, access to monitoring zones and monitoring was done with wire-line tools and equipment manufactured by Westbay Instruments Ltd. The monitoring equipment was sophisticated and equipment operators required specialized training. The instrument was installed with two different types of access ports for every zone, and the instrument components were arranged in such a way that there was up to one zone every 2 m. These characteristics allowed flexibility in the testing, sampling, and measurement activities taking place in ML-16.

Access to the zones of the Waterloo instrument was gained by placing probes into the polyethylene riser tubes. The equipment for testing and sampling was simple, easy to operate, and available from several suppliers. Two constraints of the Waterloo instrument were (1) the small diameter of the monitoring zone access tubes and (2) the fact that only six monitoring zones could be installed in a single 76-mm-diameter borehole. In formations with high transmissivity, however, the narrow diameter of the access tube would not constrain sampling. Also, six monitoring zones are often more than adequate for a single monitoring borehole.

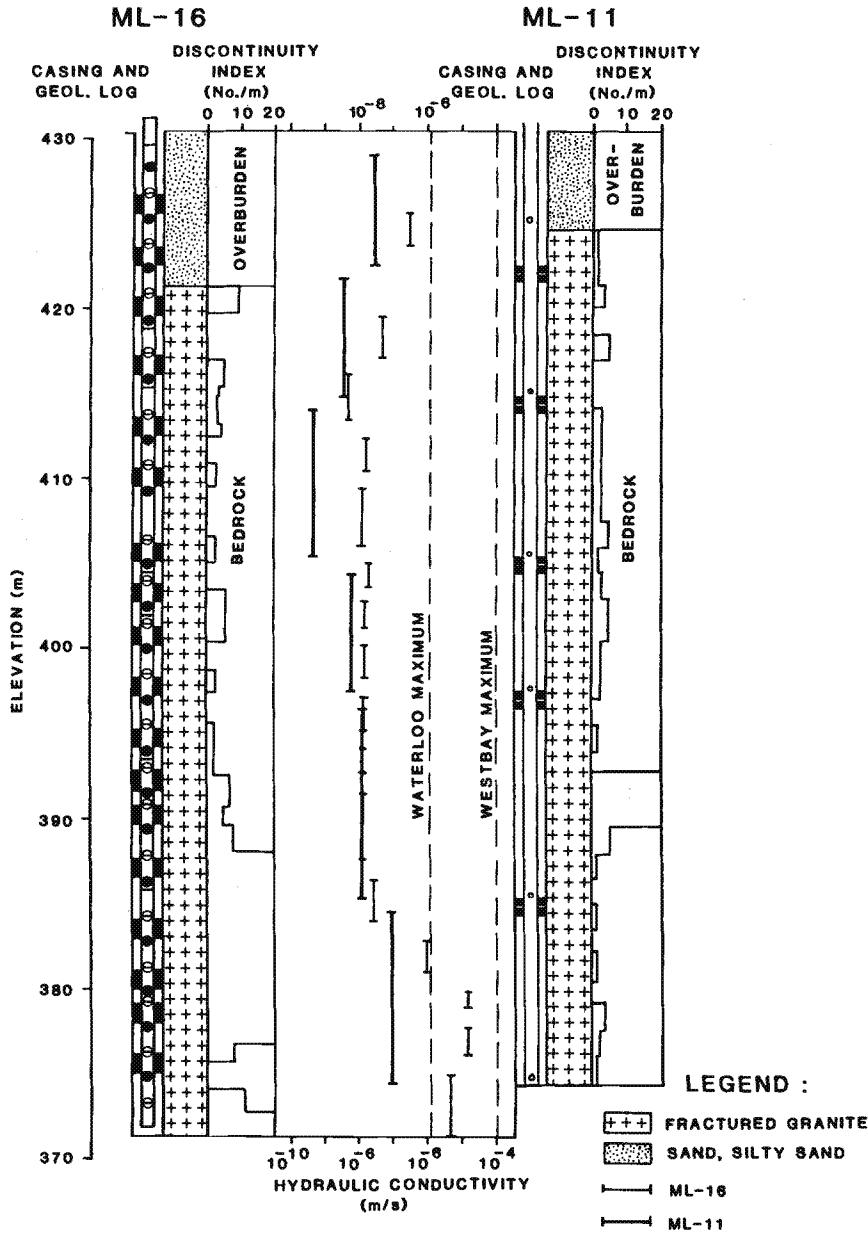


FIG. 7—Hydraulic conductivity test results for ML-16 and ML-11.

At the time this paper was prepared, the deepest known Waterloo instrument was installed to about 150 m, while Westbay instruments have been installed to depths greater than 1000 m. The deep Westbay installations contain more stainless steel parts, such as couplings, valves, and packers than were used in the installation for this study.

TABLE 3—Comparison summary for the Westbay and Waterloo instruments.

WESTBAY	WATERLOO
1. Well Design	
<ul style="list-style-type: none"> -isolated many zones in a single well -modular components allowed design changes during installation -single access tube, valved couplings for access to each monitoring zone -PVC and ABS construction with some stainless steel parts -one set of monitoring tools was used for many wells -1000-m depth capability 	<ul style="list-style-type: none"> -isolated 6 zones in a 76-mm well -modular components allowed some design changes during installation -multiple-access tubes, each dedicated to a monitoring zone -PVC construction -one set of monitoring tools was used for many wells -150-m depth capability
2. Annulus Seals	
<ul style="list-style-type: none"> -hydraulically inflated packers -inflation of individual packers by injection of water through a valve into the packer gland -deflation of packers is possible 	<ul style="list-style-type: none"> -chemically expanding packers -inflation of all packers simultaneously by adding water to interior of instrument casing -deflation of packers is not possible
3. Installation	
<ul style="list-style-type: none"> -simple layout and installation procedures -short time to install instrument string, inflating the packers took additional time -heated enclosure required in winter 	<ul style="list-style-type: none"> -simple layout and installation procedures -short installation time, more labour-intensive -packers were self-inflating, additional time to confirm packers were inflated -heated enclosure required in winter
4. Fluid-Pressure Monitoring	
<ul style="list-style-type: none"> -transducer probe and transducer control unit were used -longer time was required for readings -transducer was calibrated during monitoring routine -access tube was not hydraulically connected to the monitoring zone, freezing cannot occur -short monitoring zones may have allowed more accurate fluid pressure readings of discrete geological features 	<ul style="list-style-type: none"> -conventional electric water-level sounder was used -shorter time was required for readings -sounder was calibrated against a steel tape -dedicated access tubes are open and did freeze in winter -longer monitoring zones may have caused averaging of fluid pressure readings

TABLE 3—Continued.

WESTBAY	WATERLOO
5. Fluid Sampling -sample collection was complex -purging was simple, rapid; may not be required after first sampling -sampling tools were complicated and sampling process was labour-intensive -sampling tool decontaminated by rinsing -samples can be collected at formation pressure -winter sampling required a heated enclosure -purged 7.5 monitoring-zone volumes	-sample collection was simple when water levels were above the suction lift -purging was time-consuming since monitoring zones had low transmissivity and yield. Also, purging was required before a new sample could be collected -conventional probes and monitoring equipment -pump lines decontaminated by rinsing -a new tool is available to collect samples at formation pressure -purged 0.1 to 1.0 monitoring-zone volumes
6. Hydraulic Conductivity Tests -test method was simple, rapid -rising head test by applying rapid head change -measured the recovery of water level in the casing -maximum measurable hydraulic conductivity 1×10^{-4} m/s using liquid level techniques	-test method was simple, rapid -rising head test by removing slug from riser tube -measured the recovery of water in the riser tube -maximum measurable hydraulic conductivity 2×10^{-6} m/s using liquid level techniques
7. Quality Control Checks -couplings had serial numbers -packer inflation pressure was recorded for each packer -tests for annulus seal verification are part of the monitoring routine -checks possible on depth of downhole components after installation	-no serial numbers since prototype system -packer inflation was checked tests to verify the annulus seal can be included in the monitoring routine -checks possible on depths of downhole components after installation
8. Cost -cost of downhole components several thousands of dollars -two days for installation, less labour required -probes more than \$20k, but can be rented or leased -operating cost varied with the test activity	-cost of downhole components several thousands of dollars -one day for installation more labour required -probes less than \$2k, but optional probes and equipment are extra -operating costs varied with the test activity

Annulus Seals

The Westbay packers were individually inflated using a hydraulic inflation tool. This was a controlled operation and, hence, packer inflation pressure data were collected. After a few years of service, if needed, the packers can be deflated, and the instrument string can be removed and replaced. However, packer deflation is destructive and all the packers would have to be replaced.

The Waterloo packers were chemically inflated and formed a tight seal against the borehole wall. The inflation was confirmed by monitoring and testing the water levels in the access tubes and the instrument casing to ensure that there were no leaks or interzone connections. The borehole diameter, however, could have an effect on the sealing capability of a packer. For example, if a packer had been located at a point in the borehole where the diameter was enlarged, the sealing pressure might have been reduced. Second, if a packer in the instrument string system had failed, deflation of the other packers would not have been possible since all the packers were inflated simultaneously. Removal of the string for repair of a packer would destroy the remaining packers and might destroy other components of the instrument.

Installation

The Westbay system was installed with the assistance of a casing log. Many parts were preassembled and laid out on the ground next to the borehole. Next, the instrument string was installed and packer inflation was done on the following day. The Waterloo string was installed using similar techniques, but no wire-line tools or extra time were required to inflate the packers. Because of the winter conditions, a heated enclosure was needed for both installations, and the Westbay packer inflation equipment had to be protected from freezing. Care was required when these instrument strings were installed. Errors in the assembly and installation of the components could potentially cause problems when the instruments were to be monitored.

Fluid-Pressure Monitoring

Fluid-pressure monitoring of the Westbay instrument was done by lowering a transducer probe to a valved coupling and reading a transducer control unit, while a water-level sounder was used to monitor the Waterloo instrument. The pressure transducer and the sounder were of similar accuracy. The pressure transducer was calibrated as part of the monitoring routine. Similarly, the sounder was routinely checked and calibrated. Routine tests for zone interconnectivity should be done with both instruments. This test was part of the Westbay monitoring routine and may be instituted as part of the Waterloo routine. Readings were easier and faster with the Waterloo instrument and the equipment operator required less training.

A higher percentage of annular sealing may improve pressure monitoring detail in a complex hydrogeological environment [7]. As installed, about 28% of the Westbay borehole and 7% of the Waterloo borehole were sealed. The discrepancy in pressure readings from the two instruments may be due in part to the differences in sealing lengths. However, during the installation of ML-11, the percentage of annular sealing could have been increased simply by adding more chemical packers.

The length of the monitoring zones can also affect a piezometric pressure profile [8]. The Westbay instrument was installed with much shorter monitoring zones than the Waterloo instrument. The short monitoring zones in ML-16 may have allowed more accurate fluid

pressure readings of discrete geological features, while the longer zones in ML-11 may have caused averaging of pressure readings.

The length, density, and flexibility of the monitoring zones in the Waterloo instrument could have been improved by three different methods. First, the monitoring zones could have been shortened by adding more packers. Second, a larger-diameter hole could have been drilled to accommodate a larger casing holding more riser tubes. Finally, two holes, a short one and a longer one, could have been drilled, allowing more flexibility in locating zones.

Fluid Sampling

Fluid sampling using the Westbay instrument required the use and operation of tools and probes. For development and purging, a wire-line tool was used to open the valved pumping port coupling, and the water was removed using a pump attached to a suction line. When development and purging were completed, the pumping port valves were closed and the measurement port couplings were used for sampling the monitoring zones. No further development was required for subsequent sampling of the monitoring zones.

Samples were collected from the Westbay instrument monitoring zones at atmospheric pressure using a probe and sample bottle system. The system was decontaminated with dilute acid and deionized water rinses, followed by a formation water rinse. Depending on the procedure, samples may be collected either at the formation pressure or at the atmospheric pressure. Samples collected at formation pressure can be analyzed for the various dissolved gases.

Sampling at the Waterloo instrument was easily carried out with conventional equipment. Water samples were removed from the access tubes using a suction tube attached to a peristaltic pump. Several monitoring zones can be pumped synchronously when additional units are added to the pump. Sampling of ML-11, however, was time-consuming and the instrument could not be properly purged. This may have been due to the combined effects of the narrow port diameter, low yield, and low transmissivity of the formation.

Samples were collected from the Waterloo instrument at the water discharge line from the peristaltic pump. Prior to sampling, pumping was done at each zone until steady-state ground-water chemistry was achieved. This technique was suitable for most chemical analyses but not for dissolved gas sampling, which required collection of the sample at formation pressure. A downhole canister sampler, described by Johnson et al. [9], can be used to collect a sample at formation pressure. Also, sampling below the suction lift can be accomplished using a small-diameter gas-drive unit, described by Robin et al. [10].

Winter sampling or pressure monitoring in both instruments required a heated enclosure, such as an ice fishing tent. During winter, water in the casing annulus of the Waterloo instrument was drawn down below the frost line. However, the water in the access tubes could not be drawn down and had to be thawed prior to monitoring. The Westbay instrument did not require thawing since the water in the casing annulus was isolated from the monitoring zones and was drawn down below the frost line. Recently, modifications have been made to the commercial version of the Waterloo instrument to allow winter sampling and more flexibility in other sampling and pressure-monitoring activities.

Hydraulic Conductivity Tests

Rising-head hydraulic conductivity tests were simple and rapid with both instruments. Tests in the Westbay instrument required the use of downhole tools, while tests in the Waterloo instrument were done with conventional equipment. In the Westbay instrument,

the water level was drawn down, a tool was used to open the monitoring zone port quickly, and the rate of water level recovery was recorded. In the Waterloo instrument, the water level was drawn down with a peristaltic pump, the pump suction line was removed, and the rate of water level recovery was recorded.

The size of the entry port area and the access tube limits the maximum hydraulic conductivity that can be measured using these two instruments. When using a water-level tape, the maximum measurable limit was calculated⁶ to be about 2×10^{-6} m/s for the Waterloo instrument and about 1×10^{-4} m/s for the Westbay instrument. These measurement limits are shown in Fig. 7.

Quality Control Checks

Quality control records were generated during installation and operation of Westbay instrument components. First, the serial numbers of all of the couplings were checked and recorded so they could be traced back to the manufacturer. Second, a leak test can be performed on each port and joint prior to placement of these components into a borehole. However, this test was not performed on the installed instrument. Third, during the packer inflation operation, the volume of water injected into each packer and the final inflation pressure were recorded. After the packers were inflated, the depths of all couplings were checked against the instrument log using downhole tools. Finally, routine checks of the annular seals and casing integrity were conducted as part of the monitoring routine.

Quality control records were also generated during the installation and operation of the Waterloo instrument. The depths of the measurement ports were sounded through the access tubes. The packer inflation and casing string integrity were checked by withdrawal of water from a zone and then monitoring any effects on the water levels in the adjacent zones or the casing annulus. A review of the water-level monitoring results also gave an indication of the packer inflation and casing string integrity. There were no numerical data generated on the packer inflation pressure since the Waterloo packers are self-inflating. Finally, there were no serial numbers recorded for access ports or other key instrument components since the installed instrument was a prototype.

Costs

Compared with a conventional single standpipe piezometer, the cost⁷ for the downhole components of the two systems used was high. The cost for either system would be several thousands of dollars to equip a 60-m-deep hole with six monitoring zones. The main advantage over a single standpipe is that only one borehole is required for multiple-level information, therefore lowering the total cost, including drilling of the boreholes.

The installation costs for both systems were about the same. The instrument strings were installed in about the same amount of time, although more people were required for the Waterloo installation. An extra day was needed to inflate the Westbay packers individually, while an extra hour was needed to confirm that the packers in the Waterloo instrument were properly inflated.

⁶ The calculation is based on a variable-head hydraulic test. The following assumptions are made: a recovery of 5 m of water, a maximum recovery of one reading per 15 s, and five readings used to define the recovery response. Thus, a minimum test duration of 75 s was needed.

⁷ The cost comparison is based partly on data supplied by Westbay Instruments Ltd. and Solinst Canada Ltd.

The monitoring probes, which were required for both instruments, added to the cost. A complete set of Westbay probes was more than \$20 000, but these probes can be rented or leased. The Waterloo instrument was monitored with conventional equipment at a cost of less than \$2000; however, optional equipment and probes for the commercial version of the Waterloo instrument could increase the cost of monitoring to several thousands of dollars. The advantage in the acquisition of probes for either instrument is that they are portable and can be used to monitor many instruments in a network or at other sites.

Operating costs differ depending on the activities. Hydraulic conductivity testing took less time per zone with the Westbay system. Pressure readings were more time-consuming in the Westbay instrument. The time needed to develop the monitoring zones of the Waterloo instrument far exceeded the time needed to develop those of the Westbay instrument. However, the extra time needed for the Waterloo instrument was partly due to the low transmissivity of the formation at the test site.

Conclusions

These two multiple-level ground-water systems have completely different designs and operating approaches. The systems differ in cost, design, installation techniques, testing methods, complexity, and flexibility. Both systems were easily installed and used reliably to evaluate the ground-water chemistry, permeability, and fluid pressure at different depths within a single borehole.

These multiple-level systems have many advantages over conventional standpipe piezometers or open-borehole testing techniques. First, the cost of information per monitored zone may be lower than that for several standpipe installations, particularly when deep boreholes are required. Second, a variety of quality control tests can be performed to confirm the annular seals and check the integrity of downhole components. Finally, routine monitoring generates data on the performance of the instrument as well as the required ground-water data.

The capabilities, complexity, and cost of these or other ground-water monitoring systems should be evaluated to select the most appropriate instrument for an application.⁸ Table 3 may provide some assistance in the selection process. When evaluating an instrument, the performance aspects should be considered in the context of the overall technical requirements and objectives of a project.

References

- [1] Pearson, R., "An Overview of Hydrogeological Field Research in the Canadian Nuclear Fuel Waste Management Program," *Proceedings, International Groundwater Symposium, International Association of Hydrogeologists, Canadian Chapter, Montreal, Quebec, Canada, May 1984.*
- [2] Underwood McLellan Ltd., "Hydrogeologic Investigations, Research Area 4, Atikokan, Ontario," unpublished AECL report, available from Applied Geoscience Branch, Whiteshell Nuclear Research Establishment, Pinawa, Manitoba, Canada, 1985.
- [3] Deere, D. U., "Geological Considerations," *Rock Mechanics in Engineering Practice*, K. G. Staggs and O. C. Zienkiewicz, Eds., Wiley, New York, 1968, pp. 1-20.
- [4] Cherry, J. A. and Johnson, P. E., "A Multilevel Device for Monitoring in Fracture Rock," *Ground Water Monitoring Review*, Vol. 2, No. 3, 1982, pp. 41-44.
- [5] Black, W. H., Smith, H. R., and Patton, F. D., "Multiple-Level Ground Water Monitoring with

⁸ Note that the instruments are available through the following companies. For the Westbay instrument: Westbay Instruments Ltd., 507 East Third Street, North Vancouver, British Columbia, Canada V7L 1G4. For the Solinst instrument: Solinst Canada Ltd., Williams Mill, 515 Main Street, Glen Williams, Ontario, Canada L7G 3S9.

- the MP System," *Proceedings, NWWA-AGU Conference on Surface and Borehole Geophysical Methods and Ground-Water Instrumentation*, Denver, CO, 15–17 Oct. 1986, pp. 41–61.
- [6] Papadopoulos, S. S., Bredehoeft, J. D., and Cooper, H. H., "On the Analyses of "Slug Test" Data," *Water Resources Research*, Vol. 9, No. 4, August 1973, pp. 1087–1089.
- [7] Patton, F. D. and Smith, H. R., "Design Considerations and the Quality of Data from Multiple-Level Groundwater Monitoring Wells," *Ground-Water Contamination: Field Methods, ASTM STP 963*, American Society for Testing and Materials, Philadelphia, 1988.
- [8] Patton, F. D., "Groundwater Instrumentation for Determining the Effect of Minor Geologic Details on Engineering Projects," *The Art and Science of Geotechnical Engineering of the Dawn of the Twenty-First Century*, a volume honoring Ralph B. Peck, University of Illinois, Urbana, IL, 1989, pp. 73–95.
- [9] Johnson, R. L., Pankow, J. F., and Cherry, J. A., "Design of a Ground-Water Sampler for Collecting Volatile Organics and Dissolved Gases in Small-Diameter Wells," *Ground Water*, Vol. 25, No. 4, July–August 1987, pp. 448–454.
- [10] Robin, M. J. L., Dytynshyn, D. J., and Sweeney, S. J., "Two Gas-Drive Sampling Devices," *Ground Water Monitoring Review*, Vol. 2, No. 1, 1982, pp. 63–66.

Total Versus Dissolved Metals: Implications for Preservation and Filtration

REFERENCE: Pennino, J. D., "Total Versus Dissolved Metals: Implications for Preservation and Filtration," *Ground Water and Vadose Zone Monitoring, ASTM STP 1053*, D. M. Nielsen and A. I. Johnson, Eds., American Society for Testing and Materials, Philadelphia, 1990, pp. 238–246.

ABSTRACT: Review of ground-water quality data obtained from wells in southwestern Ohio revealed the presence of lead, cadmium, and other heavy metals. Often, no pollution source could be discovered to account for the presence of the heavy metals. In order to establish whether sampling methods were a possible source of contamination, three possible sources of spurious heavy metal values were investigated: (1) the sample preservative, (2) the sampling equipment, and (3) suspended matter in the samples.

The results of this study indicate that painted labels on nitric acid preservative ampules, used during the period 1978 to 1980, contributed to false indications of ground-water pollution. Tests of galvanized steel and polyvinyl chloride bailers showed that this equipment was not a source of lead or cadmium. Statistical analysis of a random sample of ground-water analyses from 70 wells revealed a significant relationship between sediment and lead values. Furthermore, lead values generally increased as the amount of sediment in the sample increased. This relationship probably reflects the presence of lead in particulate matter in the water sample.

Particulate matter found in well water samples can be pieces of corroded well casing or fine sediment from the formation in which the well is screened. Since very little information is available on heavy metals in Ohio's glacial deposits, additional studies of these materials should be done to provide background data for the evaluation of heavy metals in ground water.

KEY WORDS: total metals, dissolved metals, ground water, heavy metals, sample filtration, sample preservation, bailer, arsenic, cadmium, lead, iron, manganese, aluminum, chromium, copper, nickel, titanium, zinc

In the past two decades, improvements in analytical techniques have made it possible to detect heavy metals in concentrations of a few parts per billion. During this same period, many states have designated an agency to collect ground-water quality data and maintain records of these data. In most states the data represent the first analyses of ground water for heavy metals, and it is often not known what background levels (natural concentrations) of heavy metals are normal in the ground water or what levels may be due to the activities of man.

Among workers in the ground-water field, there is uncertainty over what constitutes a representative sample. Since background heavy metal concentrations are usually not known, and since these data may be questionable because of undocumented quality control and turbidity in the samples [1], it is important to evaluate existing heavy metal data carefully.

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The Ohio Environmental Protection Agency (OEPA) has maintained a ground-water quality monitoring program for both background water quality and pollution effects since its inception. Arsenic, cadmium, chromium, and lead have been monitored and detected in concentrations ranging from less than 1 ppb (the lowest detection level of the available instruments) to several times the maximum contaminant levels allowed in drinking water by the interim primary drinking water standards [2]. Lead, cadmium, and other heavy metals were detected in background monitor wells, landfill and industrial waste disposal site monitor wells, and residential wells. In many cases no obvious source could be found for the heavy metals detected in the ground water. Over a period of eleven years a considerable number of questionable analyses had been accumulated in the OEPA files.

A study was initiated with the objective of determining the validity of the heavy metal data. This study is especially important since some of the questionable data were obtained from wells used for drinking water. Another objective of the study was to evaluate sampling procedures and other conditions which could cause false heavy metal values to appear in the data. Since lead and cadmium are the heavy metal parameters that are most often unexplained in the data, this study deals mainly with them.

The data used for this study were generated by experiment or were selected from the OEPA files. The data consisted of analyses of samples from wells at facilities or areas where heavy metal pollution was unlikely. Therefore, the occurrence of lead or cadmium found in these wells was probably the result of contamination from the sampling procedure, the sample handling, the well casing, the formation in which the well was screened, or laboratory error. The study described in this paper was designed to investigate the sampling equipment, sample preservation, and formation sediment in the sample.

General Sampling Procedures

Ground-water samples were collected twice a year from all of the regular monitoring stations. Samples were obtained using a bailer (steel or plastic), pitcher pump, or portable submersible pump or from a tap using the facility's or homeowner's turbine, submersible, or jet pump. The well was usually bailed or pumped until at least one casing volume was removed. Household wells were pumped until both the casing and pressure tank volume were removed. When the casing volume was not known, the well was pumped until the water temperature reached the normal ground-water temperature for the locality. If the well water was cloudy, pumping or bailing was continued until the water became as clear as possible before the sample was collected. It was necessary to obtain a clear sample since there was no field filtration equipment available in the ground-water program at that time.

The samples for metal analyses were collected in disposable plastic bottles that had been rinsed with some of the sample water before being filled. Samples for metal analysis were preserved with nitric acid to obtain a pH of 2 or less [3]. The preservatives were furnished in 1.5 or 5-mL glass ampules.² The samples were kept on ice or in a refrigerator during storage and transportation to the Ohio Department of Health (ODH) Laboratories. At the laboratory the samples were analyzed according to U.S. Environmental Protection Agency (USEPA) procedures [3].

² The preservative ampules used by the OEPA during the period of this study were made by Poly Research Corp., New York, NY. Mention of the manufacturer's name does not necessarily constitute endorsement.

Evaluation of Nitric Acid Preservative

Introduction

From late 1978 to early 1980 many wells in the monitoring program exhibited abnormally high lead concentrations. These included ambient monitor wells and wells installed to monitor ground-water pollution. In 1979, chemists at the ODH laboratories and at the U.S. Geological Survey were concerned that the paint used to label the acid ampules and to coat the neck of the ampule might contain lead and cadmium. The ampules break apart at the neck which is etched or scratched to facilitate separation. The preservation technique involved snapping the neck of the ampule and shaking or tapping the acid into the sample bottle. Some of the paint was scraped from the ampule and tested at the ODH laboratory and found to contain lead at several hundred parts per million. However, no tests of the acid itself or of a white film found on the outside of many of the ampules were performed. This white film appeared to be a precipitate which formed on the ampules during transportation and storage. If any of the ampules were broken, acid vapors from the broken ampule seemed to cause precipitation of substances in the atmosphere onto other ampules in the shipping carton. This precipitate appeared as a white film on the ampules. Further investigation was needed to determine whether use of the preservative ampules was resulting in lead contamination of the samples.

Investigation Procedure

Several nitric acid ampules with the painted labels and necks were obtained. These ampules were divided into three groups. The first group was used to test the paint. The second group was used to test the white film on the ampules. The third group was tested for the presence of metals in the acid itself. A sample of distilled water treated with 1.5 mL of nitric acid from an unpainted batch of ampules (furnished by the same manufacturer) was used as a control.

Ampules from the first group were wiped and rinsed clean. Three of these ampules were then placed in 500 mL of distilled water which had been treated with 1.5 mL of nitric acid from an unpainted batch of ampules that had also been used for the acid in the control sample. After 5 to 7 min of immersion in the distilled water, the ampules were removed and the water was poured into a rinsed sample bottle.

The procedure was repeated for the second group, except that the outside of the ampules was not cleaned so that the white film would be exposed to the acidified water. One ampule from the third group was cleaned, opened, and added to a sample bottle filled with distilled water. The four samples, including the control, were sent to ODH laboratories for analysis.

Results of the Investigation

Table 1 shows the results of the experiment. In addition to lead and cadmium, the metals aluminum, arsenic, chromium, and copper were also tested for. The control sample did not show the presence of any of the metals at the detection levels listed in Table 1. The lead concentration in the sample from the cleaned ampule was 130 $\mu\text{g/L}$, and the cadmium concentration was 8 $\mu\text{g/L}$. The uncleaned ampules contained 140 $\mu\text{g/L}$ of lead and 16 $\mu\text{g/L}$ of cadmium. In the test of the acid preservative itself, the concentrations of all metals were below the detection limit levels. The experiment indicated that both the paint and the white film were sources of lead and cadmium. The painted ampules with a white film appeared to be a greater source of lead and cadmium. When the ampules were opened and poured into the sample container, droplets of the acid often came in contact with the paint

TABLE 1—*Nitric acid preservative study.*^a

Metal	Detection Level	Ampule Uncleaned with White Film	Ampule Cleaned, with White Film Removed	Acid Preservative	Control
Aluminum	200	* ^b	*	*	*
Arsenic	10	*	*	*	*
Cadmium	5	16	8	*	*
Chromium	30	*	*	*	*
Copper	30	*	*	*	*
Lead	5	140	130	*	*

^a All values are in micrograms per litre.

^b Asterisks indicate that the values are below detection level.

or white film, or both, on the outside edge of the ampule neck. Apparently, this was the mechanism for some of the spurious results found in previous analytical data.

Based on these results, wells which had shown high lead concentrations in 1978 and 1979 were resampled. Both lead and cadmium were found to be below the detection levels in many of these wells.

Sampling Equipment Study

Investigation Procedure

Another study was initiated to determine whether metal contamination could be introduced into the sample from the sampling equipment.

Two galvanized steel bailers (1.90 and 4.44 cm in diameter), a 10.16-cm-diameter titanium bailer, and a 10.16-cm-diameter polyvinyl chloride (PVC) plastic bailer were tested in this study. Each bailer was examined for any residues, scrubbed if necessary, rinsed once with tap water, and then rinsed with distilled water. This was the cleaning procedure followed prior to use of the bailers in the field. Each bailer was filled with distilled water and allowed to stand for 1 to 3 min to simulate field conditions, since it generally takes 1 to 3 min to retrieve a sample from a well. The sample bottles were rinsed with distilled water from the bailers to simulate field rinsing with sample water and then the sample bottle was filled with the remaining water in the bailer. A control sample of distilled water was also prepared. Each sample was preserved with 5 mL of nitric acid to obtain a pH of 2 or less. The samples were analyzed for aluminum, arsenic, cadmium, chromium, copper, iron, lead, manganese, nickel, and zinc. A sample from the titanium bailer was also tested for titanium.

Results of the Investigation

The results of analyses of the water samples from the bailers are presented in Table 2. Both galvanized steel bailers indicated aluminum concentrations at the detection limit level of 200 µg/L. The 1.90-cm-diameter bailer and the 4.44-cm-diameter bailer showed iron concentrations at 1200 µg/L and 100 µg/L, respectively. The 1.90-cm-bailer also showed manganese at the detection level of 30 µg/L. All the other parameters tested in the steel bailer samples were found to be below the detection level.

The presence of aluminum and manganese in the steel bailers is difficult to explain. These metals may be present as trace contaminants in the galvanized steel. The aluminum

TABLE 2—Bailer study results.^a

Metal	Detection Level	1.90-cm Bailer	4.44-cm Bailer	Titanium Bailer	PVC Bailer	Control
Aluminum	200	200	200	400	* ^b	*
Arsenic	10	*	*	*	*	*
Cadmium	5	*	*	*	*	*
Chromium	30	*	*	*	*	*
Copper	30	*	*	*	*	*
Iron	30	1200	100	*	*	*
Lead	5	*	*	*	*	*
Manganese	30	30	*	*	*	*
Nickel	100	*	*	*	*	*
Titanium	2000	NA ^c	NA	*	NA	NA
Zinc	30	*	*	*	*	*

^a All values are in micrograms per litre.
^b Asterisks indicate that the values are below detection level.
^c NA = not analyzed.

concentration of 400 µg/L in the titanium bailer is even more difficult to explain since that bailer was specifically designed to be free of all metals except titanium and steel in the ball valve. No aluminum or manganese was detected in the sample of water from the PVC bailer or from the control sample. The presence of aluminum and manganese was attributed to traces of these elements in the metals used to manufacture the bailers or to experimental error. No other metals were detected in the sample from the titanium bailer. All of the parameters tested were below the detection levels for the water sample from the PVC bailer. The control sample also showed all concentrations below the detection levels.

Iron appeared to be the only significant contaminant in the water sample from the galvanized steel bailers. No lead, arsenic, cadmium, chromium, copper, nickel, or zinc were detected in this portion of the study. Therefore, it was unlikely that the bailers were a source of these metals.

Sediment Study

Introduction

Another objective of this study was to determine whether suspended matter, noted in many of the samples, contained lead and other heavy metals. As previously discussed, the samples were usually preserved in the field without filtering. Since some wells would not produce a clear sample, cloudy samples were occasionally obtained. However, acid preservation of the sample could leach metals from formation sediments or other suspended material in the sample, thereby contributing dissolved metals to the water sample.

Most of the wells used for background water-quality data were municipal wells equipped with turbine or submersible pumps. Wells installed for monitoring pollution typically consisted of galvanized steel casings with brass or steel drive points, plastic screens and casings, or black iron casings with preforated sections as screens. Samples were obtained from these wells using steel or plastic bailers, a hand pump (pitcher pump), a portable 30.3 to 37.9-L/min submersible pump, or existing pumps installed by the property owner or facility operator (usually a submersible pump).

Investigation Procedure and Results of the Investigation

Seventy wells were selected for this part of the study. The analyses from these wells were selected from data for the period 1980 to 1982. Data from 1978 through 1980 were not used because of the preservative problem described above. The wells were selected from monitoring stations where there were no known sources of lead other than the well itself or the formation material. The wells included both shallow and deep wells with various diameters and casing materials (plastic, iron, and galvanized steel). All of the wells were screened in glacial deposits.

One to five analyses were selected from OEPA files for each of the 70 wells (a total of 150 analyses). The analytical reports included the lead concentrations, a pollution code, the type of well, the condition of the sample, and the identification of the well. The pollution code is a subjective term indicating whether or not the well is considered to be contaminated by some pollution source. The sample condition is a coded value judgement signifying that the sample was observed to be either clear or cloudy when it was collected.

The Statistical Analysis System (SAS) computer package [4,5] was chosen to analyze the data. The data were entered into the IBM 3031 computer at Wright State University, in Dayton, Ohio. The first step in the analysis was to select randomly one analysis for each of the 70 wells. This was done with a random selection program [4]. The data were then simplified as follows: (a) Concentrations of lead above an arbitrary value of 15 $\mu\text{g/L}$ were considered significant. Values below the 15 $\mu\text{g/L}$ level were not used because of a greater chance of sampling and laboratory error at these lower concentrations. If the concentration was 15 $\mu\text{g/L}$ or greater, the value for lead was entered as "yes"; if not, it was entered as "no." (b) The types of wells were categorized as "metal" or "plastic." (c) The pollution code was entered as a "U" for unpolluted and a "P" for polluted. (d) The sample condition was listed as "clear" or "cloudy."

The new data set was then subjected to a log-linear model analysis performed on a contingency table [6]. This analysis procedure was selected because of the necessity of using descriptive terms to represent the data rather than numerical values. The model contained four variables: (1) LEAD (no, yes); (2) SITE (unpolluted, polluted); (3) TYPE (metal, plastic); and (4) CONDITION (clear, cloudy). The resulting best fitting log-linear model produced by the computer was

$$\log m_{ijkl} = u + u_{1(i)} + u_{2(j)} + u_{3(k)} + u_{4(l)} + u_{14(il)} + u_{23(jk)}$$

where m_{ijkl} represents the expected value of lead (absent or present) listed in a cell of the contingency table; $u_{1(i)}$, $u_{2(j)}$, $u_{3(k)}$, and $u_{4(l)}$ are computer-generated numerical values for the four variables listed above; u_{14} and u_{23} are the interactions between Variables 1 and 4 and between Variables 2 and 3, respectively. The model was operated at the 95% confidence level. The interpretation of this model indicates that Variables 1 and 4 are related and Variables 2 and 3 are weakly related, and there are no other significant associations. Hence, the presence of LEAD is related to the CONDITION of the water sample. The relatively weak relationship between SITE and TYPE is probably due to the fact that some of the wells were selected from polluted sites (although lead was not suspected as a contaminant from activities at the chosen sites) and because most of the wells at contamination sites are plastic monitor wells. Therefore, there would be an artificial correlation between sampling SITES and well TYPE.

TABLE 3—Contingency table.

Condition	Presence of Lead		Probability of Lead Presence, %
	No	Yes	
Clear	31	1	0.03
Slightly cloudy	7	3	0.30
Cloudy	7	8	0.53
Very cloudy	1	12	0.92

In order to confirm the relationships found with the log-linear model, a logit analysis was performed. The logit analysis indicated that the only factor that was significant in affecting the logit LEAD content is Variable 4, CONDITION [6,7]

$$\text{logit}_{jkl} = \log \left(\frac{m_{1jkl}}{m_{2jkl}} \right) = w + w_{4(j)}$$

Since only LEAD and CONDITION were related, the parameters SITE and TYPE were dropped from the log-linear model and a more sensitive analysis was obtained by subdividing the CONDITION parameter into four levels. These levels were clear, slightly cloudy, cloudy, and very cloudy. These parameters were based on qualitative observations made at the time the samples were collected. The contingency table resulting from this log-linear model is shown as Table 3. The numbers in the right-hand column of Table 3 represent the probability of finding lead in a sample of water under the given CONDITION.

The value of the Pearson chi-square statistic for this table is 35.713 with 3 degrees of freedom. This is a highly significant value indicating that the parameters CONDITION and LEAD are strongly associated. This means that, as cloudiness increased, the probability that the well water contained lead increased. In fact, a test of the significance for the slope of a simple linear regression on these percentages (Table 3) for the CONDITION variable rendered a highly significant result: chi-square = 35.347 with 1 degree of freedom.

The association between LEAD and CONDITION was strongly positive. For relatively clear water samples there was a small likelihood of lead in the sample; for relatively cloudy water samples there was a high likelihood that lead would be found in the water sample. Furthermore, this relationship was linear in nature. This linear relationship is illustrated in Fig. 1.

Conclusions

The results of this study indicate that several batches of nitric acid preservative used during 1978, 1979, and 1980 could have contributed to erroneous lead values in ground-water quality data. It was shown that part of this contamination could be attributed to paint and, to a lesser extent, to a white film on the outside of the preservative ampule. To minimize the possibility of such contamination, unpainted ampules should be used,³ and the ampules should be thoroughly rinsed with distilled water or some of the sample water to remove any soil or precipitates before breaking them open.

³ The previously referenced manufacturer now provides an unpainted preservative ampule.

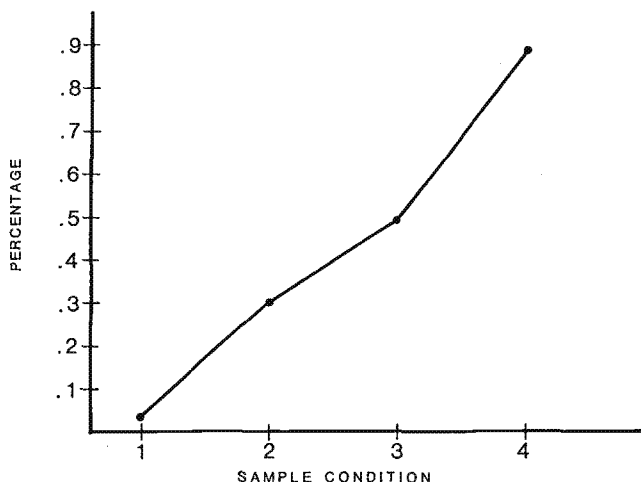


FIG. 1—Probability that the water contains lead (P) versus the sample condition.

Tests of the bailers used to collect samples showed that this equipment was not a source of lead or cadmium. Iron may have contaminated some samples collected with galvanized steel bailers, and previous iron concentration data obtained by this sampling method should be evaluated accordingly. Furthermore, these bailers should not be used where iron analysis is required.

Statistical analysis of a random sample of analyses from 70 wells revealed a significant relationship between sediment and lead values. Furthermore, lead values increased as the amount of sediment in the sample increased. This relationship probably reflects the presence of lead in the glacial deposits of southwestern Ohio. Since very little information is available on heavy metals in Ohio's glacial deposits, additional studies should be done to provide background data for the evaluation of heavy metals in ground water.

Field filtration may be necessary for samples from wells which yield cloudy water if it can be shown that any heavy metals present in the sample are contained only in the sediments which contributed to the sample's cloudiness. This may be determined by collecting both a filtered and an unfiltered sample for the metals of concern and comparing the results of analyses. Both samples must be acidified in the field, with the filtered sample acidified immediately after filtering.

It may also be advisable to analyze for metals of concern in samples of soils representative of the zone where the ground water is being sampled. This is especially valuable at a ground-water contamination investigation site. An analysis of the soils taken from a nearby area where ground-water contamination is not present will provide direct evidence of the presence of natural heavy metals in the soils. Soil samples from the vicinity of the contaminated wells can also be analyzed for total metals to obtain an indication of the metals that may have been released by the contaminated ground water and subsequently adsorbed onto the aquifer materials.

There are many sources for heavy metals in ground-water samples. This study reveals that some of the heavy metals found in ground-water samples are unrelated to pollution sources. The findings of this study substantiate the need for careful quality control measures in any ground-water monitoring program. For instance, if field blanks had been used in the late 1970s, the lead and cadmium contamination from the preservation technique would have been caught and rectified much sooner. This study has also demonstrated the

need for careful field documentation during sampling. Documentation of the sample condition provided data for a statistical analysis which led to conclusions about lead in the formation sediments.

Historical ground-water quality data is essential in the evaluation of man's impact on ground water and in developing ground-water protection programs, but such data must be carefully screened for spurious values which are the result of the sampling technique or well design rather than the actual ground-water quality.

References

- [1] Strausberg, S., "Turbidity Interferences with Accuracy in Heavy Metals Concentration," *Industrial Wastes*, March/April, 1983, pp. 20-21.
- [2] *National Interim Primary Drinking Water Regulations*, Publication No. EPA-570/9-76-003, U.S. Environmental Protection Agency, Office of Water Supply, Washington, DC, 1976.
- [3] *Methods for Chemical Analysis of Water and Wastes*, Publication No. EPA-600/4-79-020, U.S. Environmental Protection Agency, Office of Research and Development, Environmental Monitoring and Support Laboratory, Washington, DC, 1979.
- [4] Council, K. A., Ed., *SAS Applications Guide*, SAS Institute, Inc., Cary, NC, 1980.
- [5] Helwig, J. T. and Council, K. A., Eds., *SAS User's Guide*, SAS Institute, Inc., Cary, NC, 1979.
- [6] Fienberg, S. E., *The Analysis of Cross-Classified Categorical Data*, 2nd ed., MIT Press, Cambridge, MA, 1977, pp. 95-119.
- [7] Bishop, Y. M. M., "Full Contingency Tables, Logits, and Split Contingency Tables," *Biometrics*, Vol. 25, 1969, pp. 383-400.

How Flat Is Flat?—Termination of Remedial Ground-Water Pumping

REFERENCE: Spreizer, G. M., Maxim, D., Valkenburg, N., and Hauptmann, M., "How Flat Is Flat?—Termination of Remedial Ground-Water Pumping," *Ground Water and Vadose Zone Monitoring, ASTM STP 1053*, D. M. Nielsen and A. I. Johnson, Eds., American Society for Testing and Materials, Philadelphia, 1990, pp. 247–255.

ABSTRACT: Remedial actions are being initiated at many Superfund sites, and the analyses of water-quality data from these sites are being performed. The start of remediation is obvious—the system used for cleaning up is turned on; however, the determination of when this remediation is "complete" may not be clear.

With the initiation of a remedial measure, it is expected that the detected concentrations of characteristic contaminants will decrease rapidly and then level off to some asymptotic concentration. This is expected since the efficiency of remediation is generally considerably greater at higher concentrations than at lower concentrations. It is expected that some form of an exponential model, $y = C_{\text{exp}}(-kx)$, will describe the system's behavior. This is especially true when ground-water pumping is the remedial measure used, as is considered in this paper.

With Superfund remediation schemes, cleanup or performance goals are set for key compounds. The cleanup is complete when these goals are attained. However, attainment of cleanup goals may be technically infeasible in certain cases and alternative criteria may be appropriate.

This paper presents a statistical methodology, based on an anticipated exponential decrease in the contaminant concentration resulting from ground-water pumping remediation. The methodology will initially predict when remedial measures can be terminated. As the remediation program continues and additional monitoring data become available, the methodology will determine whether and when preestablished concentration limits will be attained and when the equilibrium concentration can be estimated.

KEY WORDS: ground water, statistics, termination pumping, exponential model, asymptote, empirical model, technical infeasibility, compliance waiver, SARA

SARA (the Superfund Amendments and Reauthorization Act of 1986, Public Law 99-499) requires the U.S. Environmental Protection Agency (USEPA) to use cleanup methods preferentially that reduce the volume, toxicity, and mobility of hazardous wastes at Superfund sites rather than nonpermanent remedial alternatives which only involve "containment" of contamination. Currently, bioreclamation, and pumping and treating systems (with or without caps and slurry walls) are the only two ground-water treatment methods which meet the SARA requirements for reducing the volume, toxicity, and mobility of contaminants. Bioreclamation is beginning to be used more often, but ground-water pumping and treating is still the most common method of remediating ground-water contamination.

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² President, Everest Consulting Associates, Inc.,

SARA also requires that ARARs (Applicable or Relevant or Appropriate Requirements) from federal and state laws be applied as cleanup standards at Superfund sites. In practice, this has meant that the USEPA has applied the excess cancer risk levels (for carcinogenic compounds) for the cleanup of these compounds in ground water. The 1×10^{-6} (cumulative) excess cancer risk level is a commonly used standard. Table 1 shows the 1×10^{-6} and 1×10^{-7} excess cancer risk concentrations for various carcinogens. The cumulative risk approach, which totals the risks of all compounds present, reduces the 1×10^{-6} excess cancer risk cleanup levels of individual compounds significantly. These cumulative levels are determined from the number of chemicals present and their concentrations. For instance, assuming that benzene and chloroform are present at 0.67 and 0.432 ppb, respectively (an individual 1×10^{-6} excess cancer risk level for each compound), the associated cumulative ground-water risk is 2×10^{-6} . If the cleanup standard is set at the 1×10^{-6} excess cancer risk level, then the cleanup standards will be 0.33 and 0.216 ppb for benzene and chloroform, respectively (50% of the individual cancer risk levels for each).

In addition to the analytical problems connected with detecting concentrations in the sub-parts-per-billion range, these cleanup standards are so low that they may not be achievable (i.e., with available technologies, it may not be possible to reach these standards). SARA contains provisions to address those potential situations in which compliance may be “technically impractical from an engineering perspective,” and thus, SARA permits a compliance waiver from the ARARs if infeasibility can be shown. A method for demonstrating the technical infeasibility of achieving a very low cleanup standard is presented in this paper. Emphasis will be placed on those situations in which contaminant concentrations in a ground-water pumping-and-treating system approach an equilibrium level (asymptote) that is higher than the designated ARAR concentration. When concentrations approach an asymptotic value, further real reductions in contaminant concentrations cannot be achieved in a reasonable period of time.

Background

Before remedial pumping begins, chemicals moving through aquifer materials with the natural flow of ground water are expected to be in dynamic equilibrium between the liquid and solid phases of the system. Depending on the chemical properties of the contaminants and on the chemical and physical properties of the water and the aquifer materials, the partitioning equilibrium will favor adsorption to the solid phase or solution in the liquid

TABLE 1—Excess cancer risk concentrations.^a

Compound	1×10^{-6} Excess Cancer Risk Level, $\mu\text{g/L}$	1×10^{-7} Excess Cancer Risk Level, $\mu\text{g/L}$
Benzene	0.67	0.067
Chloroform	0.432	0.0432
1,2-Dichloroethane	0.385	0.0385
1,1-Dichloroethene	0.07	0.007
Methylene chloride	5.15	0.515
Tetrachloroethene	0.77	0.077
1,1,2-Trichloroethane	0.68	0.068
Trichloroethene	3.57	0.357
Vinyl chloride	0.017	0.0017

^a Based on information and procedures in the *Superfund Public Health Evaluation Manual*.

phase. The greater a chemical's affinity for water, the longer it will spend in the liquid phase and the faster it will move.

Similarly, the greater the amount of organic material (natural and anthropogenic) in the aquifer, the slower the organic compounds will move. This is because an equilibrium condition results from the cumulative effect of the chemicals' affinities for the aquifer materials and the water. An example of this type of equilibrium and its cumulative effect is the separation of chemicals during chromatography, in which the substances with greater affinity for the solid phase are eluted more slowly (i.e., their movement through the chromatography column is retarded). Compounds have different retention (retardation) rates; these rates allow the identification of multiple compounds contained in a single sample.

When remedial pumping begins, the rate of movement of chemicals in the system increases because ground water moves faster towards the pumping well (by the influence of a greater hydraulic gradient) than it does under natural flow conditions. As the capture zone of the pumping well expands outward, larger portions of the plume or plumes and increased amounts of less contaminated water are drawn toward the pumping well. The concentrations of chemicals detected at the well will fluctuate, depending on the relative amounts of clean and contaminated water arriving at a particular time. If the pumping well is located at the leading edge of the plume, the concentration is expected to decline immediately because there will be more clean water than contaminated water in the capture zone of the pumping well. However, if the well is located in the center of the plume, the concentrations detected may increase at the start of pumping, because of the initial high concentration of the contaminants, and will decline as pumping continues.

Eventually, the capture zone will no longer expand, because water from recharge or leakage from other aquifers will balance the amount of water removed from the system through the remedial (pumping) well. At this point, a hydraulic equilibrium will have been reached. Under these new conditions, the chemical equilibrium will also have been re-established. In these situations, a portion of the total contaminant mass present in the liquid phase will be gradually removed by the remedial well. A certain proportion of the remaining mass will be associated with the solid phase and, as a result, will replenish the concentration in the ground water as contaminants are removed. The concentration of contaminants detected in ground-water samples will remain relatively constant and will approach an asymptotic level until the contaminant mass associated with the aquifer materials has been removed. The length of time that the concentration remains unchanged will depend on the initial areal distribution of the contaminant mass, the amount of the mass, and the partition coefficients of the various chemicals; this length of time can be decades for large plumes that contain high concentrations of contaminants. If the concentration approaches an asymptote, it would require an unreasonably long pumping period to achieve further real reduction in the contaminant concentration.

The exponential decrease in contaminant concentration to an asymptotic value in the ground water is similar to the time profile of a contaminant concentration in a chamber. The contaminant emissions from the surfaces in the chamber are assumed to occur at a rate E (measured in grams per second). For illustrative purposes, E is assumed to be constant; however, E can be estimated from the product of the unit emissions rate (measured in grams per square centimetre per second) and the contamination chamber surface area, A (measured in square centimetres). If G is the grams of contaminant in dissolved or entrained form in a chamber of a volume, V (in cubic metres), the average contaminant concentration, C , is then G/V (in grams per cubic metre). Water is assumed to be pumped into this chamber with a constant flow rate, Q (in cubic metres per second). It is also assumed that the incoming water contains contaminant at a concentration of C_i .

Given these assumptions, the mass balance equation is

$$\frac{dG}{dt} = E - \frac{GQ}{V} + C_i Q = E - CQ + C_i Q$$

This differential equation can be integrated and evaluated with respect to these boundary conditions:

1. C_o is the contaminant concentration at time 0, then integration of the above equation yields an expression for the concentration at time t , (C_t), and is (with intermediate integration steps given)

$$\begin{aligned}\frac{dG}{dt} &= \frac{VdC}{dt} = C_i Q - CQ + E \\ \int_{C_o}^{C_t} \frac{dC}{C_i - C + \frac{E}{Q}} &= \int_0^t \frac{Qdt}{V} \\ \ln \left(C_i - C_t + \frac{E}{Q} \right) - \ln \left(C_i - C_o + \frac{E}{Q} \right) &= \frac{-Qt}{V} \\ C_t &= \frac{E}{Q} + C_i + \left[C_o - \left(\frac{E}{Q} + C_i \right) \right] e^{-(Qt)/V}\end{aligned}$$

2. As the time t is allowed to increase, the C_t expression has the limit $E/Q + C_i$. If C_A equals $E/Q + C_i$ and Q/V is k , the equation is simplified to a form of an exponential model

$$C_t = C_A + (C_o - C_A)e^{-kt}$$

Therefore, as long as the emissions continue with rate E , the exponential model should describe the concentration in the chamber and, by comparison, the asymptotic decrease in contaminant concentrations in a ground-water pump-and-treat system.

An example of an exponential decline in the 1,1,1-trichloroethane concentration detected in samples collected from a remedial pumping well is shown in Fig. 1. After a large initial decline (about 1500 ppb) and the probable detection of a new slug of contamination (at about 400 days), the concentration detected fluctuates near the 200 ppb level, an asymptotic concentration described by graphical techniques.

Methodology

The following statistical method is used to demonstrate the potential technical infeasibility of achieving ground-water cleanup. The method is based on an anticipated exponential decrease in contaminant concentration with time. This method predicts when the remedial pumping can be terminated. As the pumping program continues and additional monitoring data become available, the method continuously revises the value of the asymptote and the time when a specified percentage of the asymptotic value is expected to be reached (an asymptote is only reached at infinity). The method is not dependent on a specific monitoring frequency; however, spatial, temporal, sampling, and analytical sources of variability must be considered before the statistical analyses are performed

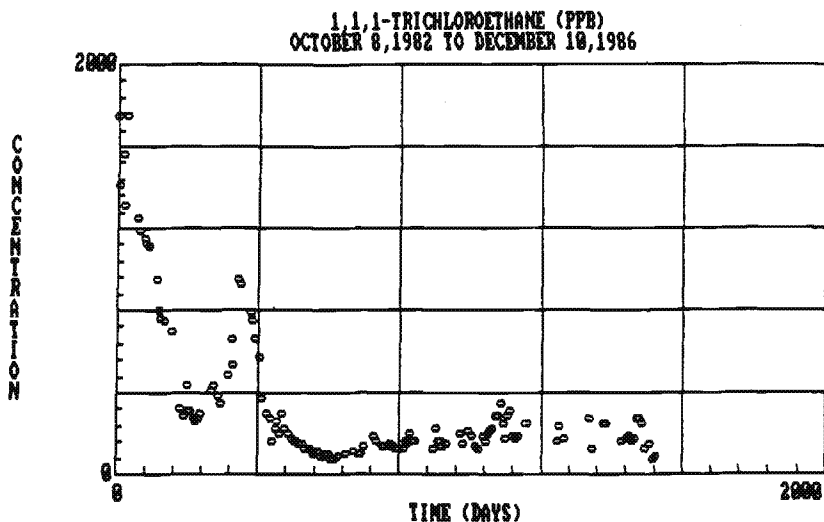


FIG. 1—1,1,1-Trichloroethane concentrations detected in samples from a remedial pumping well.

because the degree of uncertainty in conclusions drawn from the monitoring data must be assessed.

The concentrations of the indicator constituents are represented by C_{ij} , which is used to denote the concentration of constituent i , from well j , at time t . To simplify the notation utilized in the following discussion, the constituent and location subscripts have been omitted and the data are assumed to be a sequential time series of concentrations for one compound at one location. The statistical procedures described below should be followed for each indicator constituent at each well.

Given this simplification of the notation, the data are in the form C_t , where C_t is the concentration (in appropriate units) of the constituent at time t and t equals 0, 1, 2, This technical infeasibility statistical plan includes the following two steps to demonstrate that the cleanup standard may not have been achieved:

1. For an initial period, data are collected; these are summarized to monitor the progress of the ground-water remediation.
2. Upon completion of this initial collection period, the statistical procedures described below are employed to determine estimates of the time, t , (when at least a defined percentage of the asymptotic concentration has been achieved, or when the cleanup standard has been met), for each contaminant at each well.

The data are plotted (with the concentration as a function of time) and an exponential model is fitted. This exponential model is given as

$$C_t = C_A + (C_o - C_A)e^{-kt}$$

where

- C_t = concentration at time t ,
 C_A = asymptotic concentration,
 C_o = initial concentration at time 0, and
 k = decay constant

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The exponential model is expected to be a useful empirical descriptor (see the example discussed in the section on background) and should provide an adequate fit of the concentration data.

One difficulty in using a simple exponential model is that it has the property that the asymptotic value is reached only as time grows infinite. Therefore, it is necessary to define a limiting endpoint other than infinity. In this case, the endpoint is defined to be the time when the difference between the constituent concentration, C_t , and the initial concentration, C_o , is equal to a fraction, θ , of its limiting value ($C_o - C_A$). For this model, θ is defined to be 0.9, i.e., at least 90% of the asymptotic concentration. The time when this occurs, denoted t_r , is given by

$$t_r = -\ln \frac{1 - \theta}{k}$$

$$t_r = -\ln \frac{1 - 0.9}{k}$$

$$t_r = \frac{2.303}{k}$$

where

$$\theta = \frac{C_t - C_A}{C_o - C_A}$$

The value for θ must lie between 0 and 1.0. Both of these endpoints are unrealistic. The proposed value of 0.9 attempts to provide a balance between the competing, yet interrelated, objectives of constituent removal and realistic cleanup (pumping) times.

The value of k is estimated from the data. A lower confidence limit estimate of k can be used so that the upper confidence limit for t_r can be calculated using nonlinear regression methods. This provides a conservative estimate of t_r so that the remediation effort is not terminated prematurely. The following values are also calculated from the data:

- (a) estimates of the asymptotic concentration, C_A , the pumping time required, t_r , and their confidence intervals; and
- (b) statistics to measure the "goodness of fit" of the model.

A simulation of this model is given in Table 2; it assumes that the standard error of replicate observations is 10%. The first set of estimates for t_r and the upper confidence limit (UCL) for the time are made after nine years. At this time t_r is estimated as 9.5 years with the UCL indicating that it may take at least 23 years to reach 90% of the asymptotic ratio, θ ; hence, remedial pumping would continue. Note that as additional data are collected, the UCL of t_r decreases, reflecting the increased precision of this data analysis effort. After 14 years, the UCL of t_r is reduced to 12.3 years and pumping can be terminated (provided that the concentrations of other parameters have also reached 90% of the asymptotic ratio and that the UCL of t_r equals or exceeds the number of years the remedial measure has been operational).

Examples of other exponential contaminant decline models are given in Table 3. These additional forms of the exponential model may be substituted for the model discussed

TABLE 2—Simulation of the exponential model and fitting procedure.

INPUTS:			
QUANTITY	UNITS	VALUE	REMARKS
INITIAL CONCENTRATION	PPB	1000	ASSUMED FOR ILLUSTRATIVE PURPOSES
EQUILIBRIUM CONCENTRATION	PPB	150	ASSUMED FOR ILLUSTRATIVE PURPOSES
"HALF-LIFE"	YEARS	2.5	ASSUMED FOR ILLUSTRATIVE PURPOSES
CALCULATED RATE CONSTANT	1/YEAR	0.27726	CALCULATED FROM EXPONENTIAL MODEL
THETA	FRACTION	0.9	POLICY INPUT--FRACTIONAL APPROACH TO EQUILIBRIUM
REQUIRED PUMPING TIME	YEARS	8.3	TIME TO REACH THETA--TO BE DETERMINED FROM ANALYSIS
COEFFICIENT OF VARIATION	NA	0.1	RATIO OF STD. ERROR TO MEAN--USED IN SIMULATION
TIME PROFILE			
TIME YEARS	CONCENTRATION (PPB)	FRACTION REMOVABLE	FRACTION REMAINING
0	1000.0	1.00	
1	794.2	0.76	
2	638.2	0.57	
3	520.0	0.44	
4	430.4	0.33	
5	362.5	0.25	
6	311.0	0.19	
7	272.0	0.14	
8	242.5	0.11	
9	220.1	0.08	
10	203.1	0.06	
11	190.3	0.05	
12	180.5	0.04	
13	173.1	0.03	
14	167.5	0.02	
SIMULATION**			
TIME YEARS	CONCENTRATION (PPB)	THETA	COEFFICIENT OF VARIATION
0	1004.8	0.048	
1	752.8	-0.521	
2	548.4	-1.407	
3	614.7	1.822	
4	488.3	1.346	
5	377.7	0.420	
6	256.3	-1.760	
7	246.0	-0.959	
8	256.1	0.561	
9	204.3	-0.717	
10	160.5	-2.097	
11	198.7	0.443	
12	189.2	0.481	
13	134.7	-2.219	
14	196.4	1.723	
NONLINEAR FITTING			
TIME YEARS	CONCENTRATION (PPB)	THETA	COEFFICIENT OF VARIATION
0	1004.8	0.048	
1	752.8	-0.521	
2	548.4	-1.407	
3	614.7	1.822	
4	488.3	1.346	
5	377.7	0.420	
6	256.3	-1.760	
7	246.0	-0.959	
8	256.1	0.561	
9	204.3	-0.717	
10	160.5	-2.097	
11	198.7	0.443	
12	189.2	0.481	
13	134.7	-2.219	
14	196.4	1.723	
INITIAL COLLECTION PERIOD			
TIME YEARS	CONCENTRATION (PPB)	THETA	COEFFICIENT OF VARIATION
0	1004.8	0.048	
1	752.8	-0.521	
2	548.4	-1.407	
3	614.7	1.822	
4	488.3	1.346	
5	377.7	0.420	
6	256.3	-1.760	
7	246.0	-0.959	
8	256.1	0.561	
9	204.3	-0.717	
10	160.5	-2.097	
11	198.7	0.443	
12	189.2	0.481	
13	134.7	-2.219	
14	196.4	1.723	
KINETIC CONSTANT			
TIME YEARS	CONCENTRATION (PPB)	THETA	COEFFICIENT OF VARIATION
0	1004.8	0.048	
1	752.8	-0.521	
2	548.4	-1.407	
3	614.7	1.822	
4	488.3	1.346	
5	377.7	0.420	
6	256.3	-1.760	
7	246.0	-0.959	
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9	204.3	-0.717	
10	160.5	-2.097	
11	198.7	0.443	
12	189.2	0.481	
13	134.7	-2.219	
14	196.4	1.723	
LCL ESTIMATE			
TIME YEARS	CONCENTRATION (PPB)	THETA	COEFFICIENT OF VARIATION
0	1004.8	0.048	
1	752.8	-0.521	
2	548.4	-1.407	
3	614.7	1.822	
4	488.3	1.346	
5	377.7	0.420	
6	256.3	-1.760	
7	246.0	-0.959	
8	256.1	0.561	
9	204.3	-0.717	
10	160.5	-2.097	
11	198.7	0.443	
12	189.2	0.481	
13	134.7	-2.219	
14	196.4	1.723	
UCL ESTIMATE			
TIME YEARS	CONCENTRATION (PPB)	THETA	COEFFICIENT OF VARIATION
0	1004.8	0.048	
1	752.8	-0.521	
2	548.4	-1.407	
3	614.7	1.822	
4	488.3	1.346	
5	377.7	0.420	
6	256.3	-1.760	
7	246.0	-0.959	
8	256.1	0.561	
9	204.3	-0.717	
10	160.5	-2.097	
11	198.7	0.443	
12	189.2	0.481	
13	134.7	-2.219	
14	196.4	1.723	

* ESTIMATES FROM MARQUARDT'S METHOD AS IMPLEMENTED IN PLOTIT (COPYRIGHT) SOFTWARE.

** RANDOM NORMAL DEVIATES TAKEN FROM NATRELLA, M., EXPERIMENTAL STATISTICS, NBS HANDBOOK 91, NATIONAL BUREAU OF STANDARDS, WASHINGTON, D.C. 1963, TABLE A-37.

TABLE 3—Candidate contaminant decline models.

Model	Initial Value	Limit Value	Time to Reach A Fraction, Θ , of The Limit Value, t_r
$C_t = C_a + (C_o - C_a)e^{-kt}$	C_o	C_a	$t_r = \frac{\ln \left(\frac{1 - \Theta}{1 - \Theta} \right)}{-k}$
$C_t = C_a + (C_o - C_a)t^{-k}$	C_o	C_a	$t_r = \exp \left(\frac{\ln \left(\frac{1 - \Theta}{1 - k} \right)}{-k} \right)$
$C_t = a/(1 + b e^{-kt})$	$\frac{a}{(1 + b)}$	a	$t_r = \frac{-1}{k} \ln \left(\frac{1 - \Theta}{1 + b \Theta} \right)$
$C_t = C_a + (C_o - C_a)k^t$	C_o	C_a	$t_r = \frac{\ln \left(\frac{1 - \Theta}{1 - k} \right)}{\ln k}$
$C_t = kd^b t^b$	ka	k	NA

NOTE: Additional models may be employed in the analysis. Potentially useful models can be found in Frost and Pearson, or Drooper and Smith (op. cit.).

above, with the associated equations revised accordingly. All of the exponential model-fitting exercises can be done with standard curve-fitting techniques (e.g., least squares, robust regression).

Limitations of the Method

As with all analysis techniques and models, there are limitations associated with the method. The methodology discussed in this paper is simple but not naive. The only inputs required are concentration values collected over time. However, these concentrations all have uncertainties associated with the analytical results.

As discussed previously, it is necessary to determine the degree of uncertainty associated with the estimates calculated. A replicate sampling program should be undertaken to assess the sampling and analytical variability. This program should include the use of both replicate and duplicate samples and analyses. (In this usage, replication means the collection and analysis of multiple observations at a given time from one or more wells, whereas duplication is the multiple laboratory analysis of a single sample.) It may be advantageous to replicate at least selected measurements for the following reasons:

- (a) to provide a basis for selecting a useful transformation to enhance the appropriateness of the exponential model;
- (b) to permit the use of statistical tests for lack of fit of the time-series models employed; and
- (c) to optimize subsequent data collection efforts.

Estimates of the error variance components (e.g., sampling error, analytical error) are then determined. The data are analyzed using the Box and Tidwell variant of the usual analysis of variance (ANOVA) methods. These estimates are then used to characterize the reliability of the data and ultimately of the conclusions (i.e., C_A and t_r).

Conclusions

Many cleanup standards are being set at very low levels; these low levels may not be technically achievable. The method presented here, which is empirical, can be used to demonstrate that remediation has been completed even though cleanup standards have not been reached.

Flexible Selection of Statistical Discrimination Tests for Field-Monitored Data

REFERENCE: McBean, E. A. and Rovers, F. A., "Flexible Selection of Statistical Discrimination Tests for Field-Monitored Data," *Ground Water and Vadose Zone Monitoring, ASTM STP 1053*, D. M. Nielsen and A. I. Johnson, Eds., American Society for Testing and Materials, Philadelphia, 1990, pp. 256-265.

ABSTRACT: The capabilities and limitations of a number of statistical methods for statistical discrimination testing are examined. Arguments are developed for flexibility in selecting alternative statistical models. Several alternatives for including detection limit data are developed.

KEY WORDS: ground water, significance tests, *t*-test, detection limits

In ground-water studies, there is a frequent need to determine whether a significant difference exists between two or more sets of data. This requirement arises, for example, when examining whether a landfill site has impacted the ground-water quality. However, the numerous sources of variability implicit in ground-water quality measurements (e.g., sampling errors, inadequate sample storage and preservation techniques, analytical errors, and the vagaries of nature) require that statistical discrimination tests be employed.

Compounding the problems of uncertainty/variability are features of ground-water quality records including the following:

1. A number of constituents may be of concern, and laboratory costs are high, which tends to act against development of sizable monitoring records.
2. Many phenomena take years to evolve, and therefore the available time frame for sampling programs represents only a "window" of a temporally varying process.

To consider the problem of assessing the significance of the impact of such variables, a number of different statistical tests might be appropriate. Each test involves different assumptions, and there are no absolute rules stipulating which test to apply, only guidelines. Note that one must always temper statistics with a clear understanding of the problem, so that spurious information is not introduced, nor valid information omitted. Statistical analyses of data are not interpretations of the facts; the analyses are just another way of making the facts easier to see, and therefore to interpret. The various statistical methods are tools for data analysis, which, like any tools, have proper and improper applications. The intent of this paper is to argue for the flexibility that will allow the alternative statistical models to be used for a problem.

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Alternative Tests

When a finding that is very unlikely to have arisen by chance is obtained, the result is referred to as being statistically significant. In other words, the finding is difficult to ascribe to chance, and the difference must, in all common sense, be accepted as a real difference.

In response to the need to identify statistically significant differences, a number of discrimination testing procedures have been proposed. No absolute rules are available for selecting the discrimination procedures to use in a specific application, only guidelines. To a large extent, the selection of the best procedure involves careful scrutiny of the characteristics of the problem at hand and the assumptions implicit in the particular discrimination technique being considered.

t-Tests

The first of these statistical significance tests to be developed was the student's *t*-distribution. Mathematically, the *t*-test procedure, as presented by Fisher [1], allows the testing of whether the means from two sets of measurements, say *X* (where the elements of *X* are x_i , where $i = 1, 2, \dots, n_x$) and *Y* (where the elements of *Y* are y_j , where $j = 1, \dots, n_y$) are the same. Assuming that *X* and *Y* are normally distributed, with the same variance, but that their population means, μ_x and μ_y , may be different, then the difference between the sample means, $\bar{x} - \bar{y}$, will be normally distributed with the mean $(\mu_x - \mu_y)$. Then

$$t = \frac{|\bar{X} - \bar{Y}|}{\sqrt{\frac{(n_x - 1)S_x^2 + (n_y - 1)S_y^2}{n_x + n_y - 2}}} \sqrt{\frac{1}{n_x} + \frac{1}{n_y}} \quad (1)$$

where $||$ denotes the absolute value sign, S_x^2 represents the variance of sample set X_i ($S_x^2 = \Sigma(x_i - \bar{x})^2 / (n_x - 1)$), and S_y^2 represents the variance of sample set Y_j . The *t* follows a *t*-distribution with $m + n - 2$ degrees of freedom.

Noteworthy points regarding the above include the following:

- (a) the assumption that distributions of *X* and *Y* have the same variance is essential to the argument, and
- (b) the *t*-test is based on the assumption that this underlying distribution is normal or Gaussian.

Unfortunately, these assumptions are frequently violated in monitoring records.

Parametric Versus Nonparametric Tests

Out of the fundamental *t*-test derivations, a number of alternative tests for statistical discrimination have been developed. One group of tests is referred to as parametric (meaning that they utilize the magnitudes themselves), while the second class of tests is referred to as nonparametric (meaning that they use the relative rankings of the magnitudes).

The parametric tests are very good if the following obtain:

1. The assumptions of randomness, independence, and normal distribution are true.
2. An additional test, which compares the distributions, is completed and ensures that there is no significant difference in the variances at a specified level of significance.

TABLE 1—Summary of test statistics,

Test	Two Sample t-Test	Satterthwaite Approximation to the Two Sample t-Test
t statistic	$t = \frac{ \bar{X} - \bar{Y} }{\sqrt{\frac{1}{n_x} + \frac{1}{n_y} \cdot \frac{(n_x - 1)S_x^2 + (n_y - 1)S_y^2}{n_x + n_y - 2}}}$	$t_1 = \frac{ \bar{X} - \bar{Y} }{\sqrt{\frac{S_x^2}{n_x} + \frac{S_y^2}{n_y}}}$
Degrees of Freedom	$df = n_x + n_y - 2$	$df = \frac{\left(\frac{S_x^2}{n_x} + \frac{S_y^2}{n_y}\right)^2}{\left[\frac{\left(\frac{S_x^2}{n_x}\right)^2}{\frac{n_x}{n_x - 1}} + \frac{\left(\frac{S_y^2}{n_y}\right)^2}{\frac{n_y}{n_y - 1}}\right]}$
Comments:	Since σ is unknown it is replaced by S , the sample standard deviation.	Note: round 'df' down to the next nearest integer.

Nonparametric tests do not require a test on their parameters (i.e., equal variances) for these reasons:

1. The data do not need to be normally distributed, since these tests use ranks in place of the original data.
2. The data are ranked, which also tends to suppress the outliers, and thus the tests function reasonably well for highly skewed data. This paper does not provide examples of nonparametric tests. (For details, see, for example, Ref 2.)

Insofar as the parametric tests are concerned, and out of the fundamental *t*-test derivations, a number of alternative tests for statistical discrimination have been developed. The different tests include the following:

1. Modified *t*-tests have been developed (e.g., Satterthwaite's approximation (see Imam and Conover [3]) and Cochran's approximation to the Behrens-Fisher *t*-test [4]. These tests accommodate the situation of unequal variances. One should note, however, that the tests are indeed approximate, as the names of the tests indicate, and the nature of these approximations must be reflected during their utilization (see Ref 5 for further details on the nature of the approximations). The difference between Cochran's test and Satterthwaite's

degrees of freedom, and assumptions.

Cochran's Approximation
to the Behrens-Fisher Test

$$t_2 = \frac{|\bar{X} - \bar{Y}|}{\sqrt{\frac{S_x^2}{n_x} + \frac{S_y^2}{n_y}}}$$

$df_x = t$ - tables with $n_x - 1$
degrees of freedom
 $df_y = t$ - tables with $n_y - 1$
degrees of freedom

$W_x = \frac{S_x^2}{n_x}$ and $W_y = \frac{S_y^2}{n_y}$
with the result the
comparison t -statistic is

$$t_c = \frac{W_x t_x + W_y t_y}{W_x + W_y}$$

$$S_x = \sqrt{\frac{\sum (x_i - \bar{X})^2}{n_x - 1}}$$

$$S_y = \sqrt{\frac{\sum (y_i - \bar{Y})^2}{n_y - 1}}$$

Paired t -Test

$$t = \frac{\bar{D}}{\frac{S_D}{\sqrt{n_x}}}$$

where $D_i = x_i - y_i$ for $i=1 \dots n_x$

$$\text{and } S_D = \sqrt{\frac{\sum_{i=1}^{n_x} (D_i - \bar{D})^2}{n_x - 1}}$$

$df = n_x - 1$

test is strictly in calculating t_c , the critical t -statistic. Cochran's test weighs the individual t values for each sample, while Satterthwaite's test calculates new degrees of freedom.

2. Paired sample t -tests may be used when the sample populations, X and Y , are not independent. For some kinds of correlated data, it is possible to define the population being sampled so as to remove or minimize the effect on the test outcome (e.g., cyclical variations in both background and monitored data are not included in the calculation of the standard error). This is done by pairing the individual observations from the correlated populations and then testing the differences between the observations. Once the differences in the pairs are calculated, the differences are treated as a single, random, independent sample. With the pairwise test, there is one half of the degrees of freedom of the two-sample test, but the standard error calculation is smaller.

A summary of the mathematical equations implied in the above-named tests is included in Table 1.

Comparison of t -Test Versus Cochran's Approximation

One should note that both Cochran's and Satterthwaite's tests are approximate formulas for the t -test that allow relaxation of the equality of variances requirement and should be considered the most appropriate test in these situations. However, there are situations in which the assumptions implicit within the basic t -test are *not* violated, and the utilization of the approximate tests in such situations will imply different findings from those intended. This concern may be quantified, as an example, for the case where $n_x = 3$ (the downgradient location has three samples) and $n_y = 4$ (the upgradient has four samples), with the variance ratio $S_x^2/S_y^2 = 4$. The t -test allows a deviation of $|\bar{X} - \bar{Y}|$ of 2.28 before a significant difference is identified (with a level of significance of 5%, by one-tailed t -test). Translation of the allowable deviation by Cochran's test into the equivalent deviation for the t -test corresponds to $|\bar{X} - \bar{Y}| = 3.56$, which, in turn, gives a significance level of 1.3%. In other words, if Cochran's test is used instead of the t -test, the level of significance is not 0.05 but 0.013. (Concerns such as these are further described by McBean et al. [5].) Therefore, the data analyst should not blindly follow stringent guidelines without matching his or her data concerns with the various test assumptions. A summary of statistical discrimination tests in frequent use in the assessment of ground-water quality is provided in Table 2.

Consideration of Detection Limit Data

The need for flexibility in the approach to analysis is further exemplified by situations involving detection limit data. All laboratory analytical techniques have detection limits below which only "less-than" values may be recorded. The reporting of less-than values provides a degree of quantification, but, even at or near their detection limits, the concentration levels of particular contaminants may be of considerable importance because of their potential health hazards. Also, if the subsequent statistical analyses that utilize the monitoring results indicate a significant difference between two monitoring locations, a greatly increased intensity of monitoring may result. The manner in which the "less-than" information is dealt with can have a substantial impact on monitoring programs.

To examine the nature of such difficulties, consider the utilization of the parametric statistical significance testing procedures. These methods require replacement of less-than values by quantified values. In practice, a frequent approach to allow quantification involves assignment of less-than values as "equal to" either (1) the detection limit, (2) one half the detection limit, or (3) zero. These assignments provide a degree of quantification but may seriously affect subsequent utilization of the parameters used in the parametric tests. As an indication of the consequences, consider the example where the less-than values are assumed to be equal to their detection limit (Case 1, above). In this case, the estimate of the mean is high and the estimate of the standard deviation is low. Alternatively, where the less-than values are assumed to be equal to zero (Case 3, above), the estimate of the mean is relatively low and that of the standard deviation relatively high. Cases (1) and (3) indicate the bounds—if the results of the statistical significance tests are the same for Cases (1) and (3), then the assignment of the values for the less-thans does not matter; however, if the tests result in a different finding, an alternative procedure is necessary.

Results contained in the study by McBean and Rovers [6] indicate several additional alternatives. However, another approach may be of use if more than a single constituent has been monitored and there is a correlative behavior between several of the constituents. The procedure will be demonstrated by application to recent monitoring data, as listed in the second and third columns of Table 3. As indicated in the tabulated data for 1,1,2-trichloroethylene (TCE) and orthonitraniline (ONA), the TCE data contain 6 less-than

TABLE 2—Summary of statistical discrimination tests in frequent use.

Type	Test	Major Assumptions	Example Applications
Parametric	Two sample t-test	(i) the distribution of the two samples have the same variance(**), (ii) the distributions are normal or gaussian	Rovers and McBean (1981) ⁸ and McBean and Rovers (1984) ⁶
	Approximate t-tests (i) Satterthwaite's Approximation (ii) Cochran's Approximation to the Behrens-Fisher t-test (Cochran, 1964)	(i) relaxation of the assumption regarding variances, (ii) the distributions are approximately normal or gaussian. Note: This has been explicitly characterized by the coefficient of variation which must be less than unity, in RCRA (Federal Register, 1982).	McBean and Rovers (1988) ⁵
	Data-transformations(*) prior to use of the two sample t-test	(i) the distributions of transformed variables have the same variance(**); (ii) the distributions are normal or gaussian	Rovers and McBean (1981) ⁸
	Pairwise t-test - when the sample populations are differenced	(i) the differences between the two samples are distributed as normal or gaussian	
Non-parametric	Mann-Whitney test		

Notes:

(*) data transformation approaches have a degree of arbitrariness with respect to the transformation to be employed for a specific application.

(**) a prior F-test indicates whether the sample variances are sufficiently alike to warrant the assumption that they are independent estimates of the same population variance.

TABLE 3—Historical data record for ONA and TCE, in micrograms per litre.^a

Sample No.	ONA	1,1,2-TCE	1,1,2-TCE (estimated)
1	1.1	<5	6.5
2	1.8	10	10
3	0.53	<5	4.7
4	1.5	7	7
5	1.4	7	7
6	3.7	15	15
7	1.1	6	6
8	1.9	7	7
9	1.2	<5	6.8
10	0.29	<5	3.9
11	0.42	<5	4.3
12	0.9	<5	5.8
13	3.4	14	14
14	0.22	5	5

^a The data were obtained from monthly samplings for ONA and 1,1,2-TCE concentrations.

detection values from a record length of 14. Utilizing the coincident data record in which both TCE and ONA have levels greater than the detection limit, the degree of correlation between the two chemicals was quantified ($r^2 = 0.91$) and the following linear regression equation was derived.

$$1,1,2\text{-TCE} = 3.04 + 3.11 \cdot \text{ONA} \tag{2}$$

The resulting data are plotted in Fig. 1.

Equation 2 was then used to predict the values for the less-than data for 1,1,2-TCE. The resulting data set for 1,1,2-TCE is listed in the fourth column of Table 3.

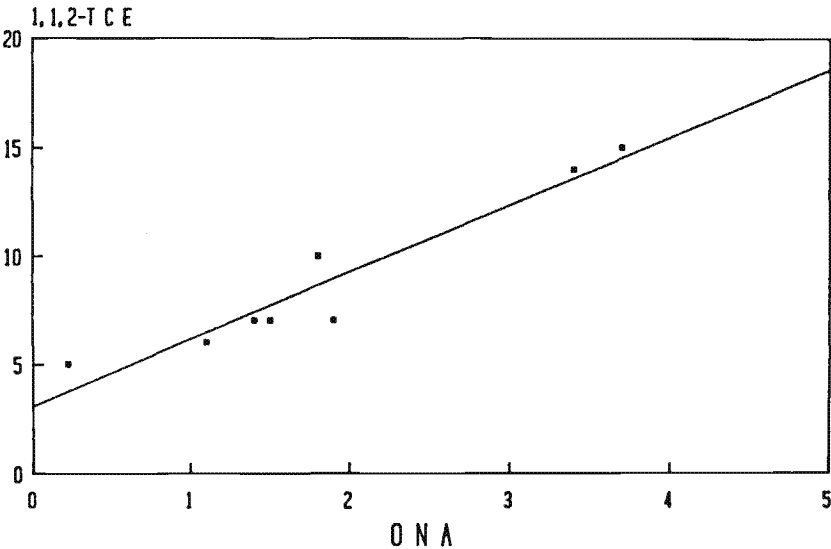


FIG. 1—ONA versus 1,1,2-TCE data with detection limit data removed. The concentrations are in micrograms per litre.

The 95% confidence interval (CI) was established (for details on the procedure, see Zar [7]) for the 1,1,2-TCE values and has been plotted, along with the estimated values for the detection limit data, in Fig. 2.

As can be seen from Fig. 2, three of the data points presented as less-thans were estimated, using the regression equation, to be greater than the detection limit. Several comments are appropriate:

The best estimate of a 1,1,2-TCE measurement, given the ONA level, is on the regression line. However, as apparent from Fig. 1, the data pairs with no less-thans reported do not lie along the regression line—there are deviations from the line. It is the purpose of the 95% confidence limits to demonstrate one measure of the degree of variability that exists.

Therefore, the best estimate of individual values is on the line, but the range of possible values (as exemplified by the confidence limits) is considerable. In particular, one should note that many of the potential values for the extrapolated values are less than 5 $\mu\text{g/L}$ for 1,1,2-TCE.

Correlation analyses can be used to extrapolate or estimate values of the less-than data. The viability of the extrapolation requires the following:

(a) high correlation between the constituents, i.e., ONA and 1,1,2-TCE data for the example at hand;

(b) an expectation of similar behavior for the constituents (this assumption is particularly defensible if the chemicals are in the same family but is also potentially useful in the case of the same forcing conditions, e.g., emanating from the same contaminant source); and

(c) if the correlation analysis is performed over temporal data, the data records of each constituent should be examined for persistence of the less-than detection limit data. A persistent trend of less-than data may indicate a disappearance of the compound as a result

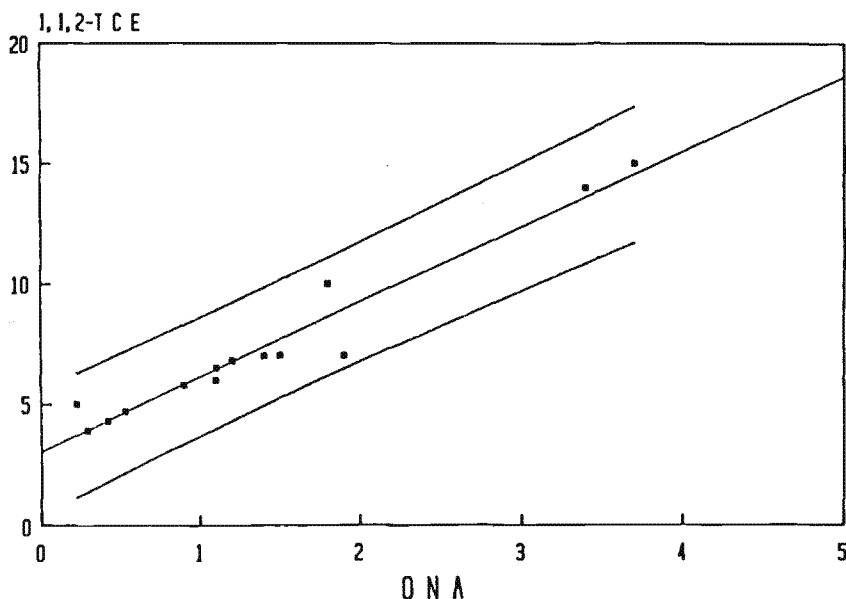


FIG. 2—ONA versus 1,1,2-TCE with 95% confidence intervals of Y estimated. The concentrations are in micrograms per litre.

TABLE 4—*Estimates of means and variances as determined by alternative methodologies.^a*

Statistical Parameter	Correlation Method	Equality Assignments			Plotting Method	
		Equal to Detection Limit	One Half Detection Limit	Zero	Normal Distribution	Lognormal Distribution
Mean	7.4	7.2	6.1	5.1	6.0	5.6
Standard deviation	3.4	3.4	4.3	5.3	6.6	6.4

^a All concentrations are in micrograms per litre.

of either natural or man-induced causes. Correlation analyses inherently assume the comparable existence or behavior of both compounds.

As a comparison to the correlation method, the mean and standard deviation of the 1,1,2-TCE data were calculated using two of the methods presented by McBean and Rovers [6]. First, the detection limit data were replaced by zero, then, by one half the detection limit, and, finally, by the detection limit. In all three cases, the mean was estimated to be lower and the standard deviation was found to be higher than the values calculated using the correlation analysis technique, as is apparent in Table 4.

The second method involved plotting data on both normal and lognormal distribution paper as per McBean and Rovers [6]. The estimated mean using this procedure equals 6.0 and 5.6 $\mu\text{g/L}$ for the normal and lognormal values, respectively, and the standard deviations were estimated to be 6.6 and 6.4 $\mu\text{g/L}$, respectively. Note that, based on the limited data set, both the normal and the lognormal plot produce a relatively straight line.

The findings on estimates of the mean and standard deviation are summarized in Table 4. A comparison of several possible tests will provide the reviewer with alternative, but justifiable, means of characterizing ground-water data.

Conclusions

The quantification by statistical analysis, in general, and hypothesis testing, in particular, is intended to reduce and summarize observed data, to present information precisely and meaningfully, and to determine some of the underlying characteristics of the observed phenomena. It should not be presumed that current knowledge of statistical analysis is always sufficient to provide answers to many of the complex problems of ground-water contamination. Nevertheless, the ability to provide answers is enhanced by a flexible approach, using a variety of alternative models, while understanding the limitations of the various models and using a considerable degree of engineering judgment.

References

- [1] Fisher, R., *Metron*, Vol. 5, 1926, p. 90.
- [2] Unwin, J., Miner, R. A., Srevers, G., and McBean, E., "Groundwater Quality Data Analysis," *NCASI Technical Bulletin*, No. 462, 1985, pp. 148-160.
- [3] Imam, R. A. and Conover, W. J., *A Modern Approach to Statistics*, Wiley, New York, 1983.
- [4] Cochran, W., "Approximate Significance Levels of the Behrens-Fisher Test," *Biometrics*, March 1964, p. 191.
- [5] McBean, E., Kompter, M., and Rovers, F., "A Critical Examination of Approximations Implicit in Cochran's Procedure," *Ground Water Monitoring Review*, Winter 1988, pp. 83-87.

- [6] McBean, E. A. and Rovers, F. A., "Alternatives for Handling Detection Limit Data in Impact Assessments," *Ground Water Monitoring Review*, Spring 1984, pp. 42-44.
- [7] Zar, J. H., *Biostatistical Analysis*, 2nd Ed., Prentice-Hall Book Co., Toronto, 1984.
- [8] Rovers, F. A. and McBean, E. A., "Significance Testing for Impact Evaluation," *Ground Water Monitoring Review*, Vol. 1, No. 2, 1981, pp. 39-43.

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In Situ Analysis of Benzene, Ethylbenzene, Toluene, and Xylenes (BTEX) Using Fiber Optics

REFERENCE: Chudyk, W., Pohlig, K., Exarhoulakos, K., Holsinger, J., and Rico, N., "*In Situ* Analysis of Benzene, Ethylbenzene, Toluene, and Xylenes (BTEX) Using Fiber Optics," *Ground Water and Vadose Zone Monitoring, ASTM STP 1053*, D. M. Nielsen and A. I. Johnson, Eds., American Society for Testing and Materials, Philadelphia, 1990, pp. 266–271.

ABSTRACT: *In situ* analysis of ground water is an attractive alternative to conventional sampling and analysis methods, since it eliminates sample handling and chain-of-custody concerns. The prototype instrument recently developed by our research team uses remote laser-induced fluorescence (RLIF) to measure aromatic organics *in situ*. The work discussed here concerns the aromatic organics fraction of gasoline (benzene, toluene, and xylenes). RLIF can theoretically detect over half of the organics on the U.S. Environmental Protection Agency's Priority Pollutants List.

Typical analysis for these contaminants involves routine repetitive analysis, such as quarterly gas chromatography/mass spectroscopy (GC/MS) on samples taken from wells. Such monitoring is labor intensive, requiring a sampling team as well as a laboratory. Samples for GC/MS often take many weeks for analysis, so that remedial action information is late in arriving. An analysis is needed that is quick, simple, and less costly, providing a look at the situation as it currently exists.

The authors propose that routine GC/MS monitoring at characterized sites be replaced with RLIF measurements, supplemented with annual GC/MS analysis as a means of validation. Since RLIF analysis is nonspecific, this approach would be sound only at sites containing known aromatics as the contaminants of interest, or at sites where the aromatics concentration correlates with other problem compounds. Comparisons of RLIF and GC sensitivity for aromatic organics are discussed, with emphasis on the relative advantages and limitations of each method.

KEY WORDS: ground-water monitoring, *in situ* measurements, laser-induced fluorescence, aromatic contaminants, gas chromatography

An interdisciplinary research team at Tufts University, in Medford, Massachusetts, has recently developed a remote laser-induced fluorescence (RLIF) technique which is capable of measuring aromatic organic ground-water contaminants *in situ*. This method uses a laser as a light source, and fiber optics to carry laser excitation light down a well and resulting fluorescent light back up to the surface to a detector [1,2]. Fluorescence methods have been used previously in the laboratory to identify and quantitate contamination of water by petroleum products [3]. Our application uses a similar approach with a portable unit appropriate for pilot or field operation.

Traditional sampling and laboratory analysis can cause changes in sample composition, because of contamination by samplers, and also changes in the sample's pH, redox potential, or loss of volatiles content. *In situ* measurements allow rapid acquisition of informa-

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tion at relatively low cost, making them attractive as an alternative method for ground-water monitoring [4].

Theoretical Background

Aromatic organics include the benzene, ethylbenzene, toluene, and xylenes fraction of petroleum fuels, as well as many other ground-water contaminants. Most aromatic organic compounds contain a benzene ring as part of their molecular structure. This part of the molecule can adsorb light of the right wavelength, typically ultraviolet, and later emit the adsorbed energy as fluorescence. The intensity and wavelength of fluorescence are functions of molecular concentration and structure, respectively. Measurement of fluorescence of aromatics dissolved in water has been used to measure their concentration and determine the type of molecule present, leading to identification of sources of petroleum product spills [3]. Such an approach has typically been performed in the laboratory with relatively large equipment. Recent developments in high-energy light sources and fiber optics have allowed us to construct and test a field-portable fluorescence-measuring instrument [1,2].

Procedure

The first prototype RLIF instrument used for measuring aromatic ground-water contaminants has been described elsewhere [2]. The latest prototype RLIF instrument has the light source, detector, and signal-processing electronics mounted in a series of portable modules connected to a fiber-optic sensor placed in a well. In pilot laboratory studies, power is provided by a conventional wall plug, while, in the field, the unit is usually powered by a 3-kW portable generator. The excitation module, containing the light source, is shown in Fig. 1. The components in the excitation module are used to generate ultraviolet light at 266 nm and to focus this excitation light into a fused-silica optical fiber. Pulsed light at 532 nm is emitted by the laser, along with some 1064-nm light. The 1064-nm filter removes the 1064-nm light, and the remaining 532-nm light passes through a frequency-doubling BBO optical crystal. Some of the 532-nm light is frequency-doubled to 266 nm, in the ultraviolet (UV) range. The dichroic mirror selectively reflects only 266-nm light, which is further steered and focused by a prism and lens into the excitation fiber. Two fibers, one excitation and one emission, are joined in the sensor, which is placed into the well to be sampled.

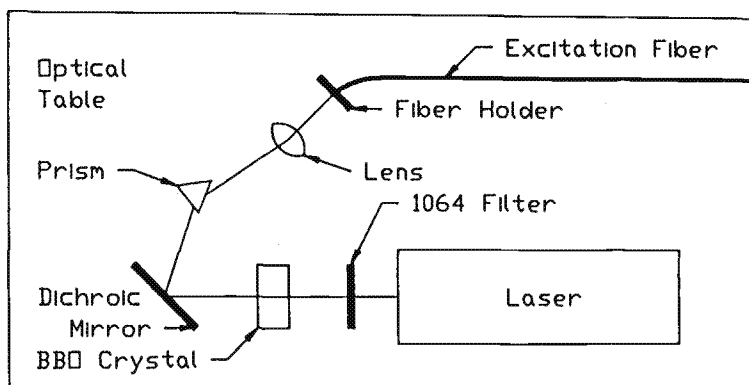


FIG. 1—Excitation module.

The sensor is constructed of fused silica glass, stainless steel, and polytetrafluoroethylene (PTFE) so as to minimize contamination of well water by the sensor. The maximum diameter of the sensor is 19 mm (¾ in.), so that it will fit virtually any monitoring well. It may be placed before the well is bailed, and the establishment of formation water can be monitored as purging occurs. For long-term or continuous monitoring, sensors can be left in place in a well. On-site calibration is typically performed with a 1-ppm phenol solution.

Light from the excitation fiber excites aromatic molecules dissolved in the water, causing some of them to fluoresce. The fluorescence is emitted from the molecules randomly in all directions, so some of it is collected by the emission fiber and led back up to the detection module. Scattered emission light is also collected by the detection fiber. The detection module is shown in Fig. 2. About 10% of the collected light is directed onto the power-normalization photomultiplier tube (PMT) to provide an internal laser-power reference signal. The remaining light is spectrally filtered to block interferences and directed into the detection PMT.

The detector module output is processed in the electronics module, which is shown in Fig. 3. Each PMT in the detector produces a current proportional to the amount of light it receives. Current from the power-normalization PMT is proportional to the amount of excitation light sent into the well, which is a function of the laser power during a given pulse. Current from the detection PMT is proportional to the fluorescence of contaminants in the water generated by the same pulse of excitation energy, measured by the power-normalization PMT. Each of the current signals from the PMTs is separately integrated electronically to yield a voltage. The ratio of these voltages is used to determine the intensity of fluorescence from contaminants in the well. The fluorescence intensity is compared with standard curves of fluorescence versus concentration for the contaminants of interest, so that the concentration can be determined. A personal computer (PC) is used for instrument control and data logging. The other parts of the electronics module are power supplies for the laser and detector module, as well as interface electronics for the PC.

Gas chromatography (GC) analysis is used as a check on instrument performance. Since all aromatic molecules are expected to fluoresce, they are all theoretically detectable using RLIF. A list of aromatics detected by U.S. Environmental Protection Agency (EPA) Method 502.2 is presented in Table 1. Since these compounds are all aromatics, they are also theoretically detectable using RLIF. Compounds already tested with RLIF (benzene, toluene, and xylenes) are denoted by an asterisk.

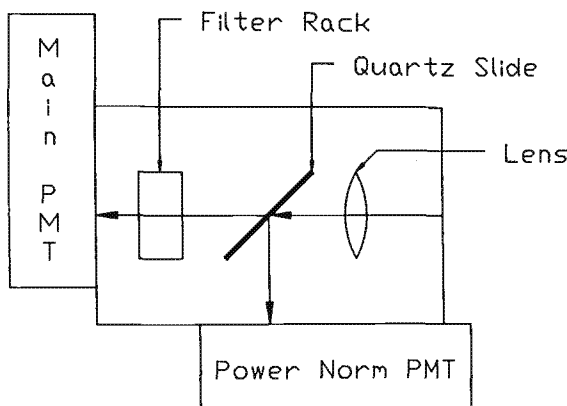


FIG. 2—Detection module.

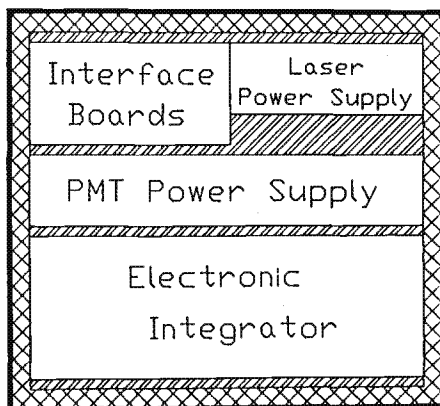


FIG. 3—Electronics module.

Results

Previous work has tested model contaminants including phenol, ortho-cresol, toluene, xylenes, and humic acid, as well as both landfill and bark-pile leachates [1,2]. The limits of detection found in the laboratory for these compounds are presented elsewhere [1] and are an indication of the instrument response. Based on those earlier data, single-ring aromatics seem to be easily detected using RLIF. Gasoline is known to contain the single-ring aromatics benzene, ethylbenzene, toluene, and xylenes (BTEX), so that analysis for BTEX

TABLE 1—Aromatics theoretically detected by both RLIF and EPA Method 502.2.

Benzene ^a
Bromobenzene
<i>n</i> -Butylbenzene
<i>sec</i> -Butylbenzene
<i>tert</i> -Butylbenzene
Chlorobenzene
2-Chlorotoluene
4-Chlorotoluene
1,2-Dichlorobenzene
1,3-Dichlorobenzene
1,4-Dichlorobenzene
Ethylbenzene
Isopropylbenzene
<i>p</i> -Isopropyltoluene
Naphthalene
<i>n</i> -Propylbenzene
Styrene
Toluene ^a
1,2,3-Trichlorobenzene
1,2,4-Trichlorobenzene
1,2,4-Trimethylbenzene
1,2,5-Trimethylbenzene
<i>o</i> -Xylene ^a
<i>m</i> -Xylene ^a
<i>p</i> -Xylene ^a

^a Compounds already examined via RLIF.

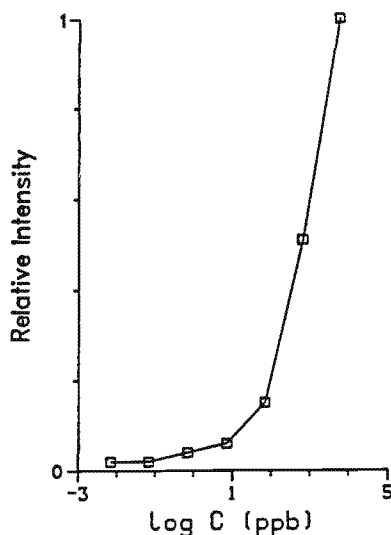


FIG. 4—RLIF response versus the gasoline concentration.

gasoline components is a natural extension of earlier work. This study focused on unleaded gasoline. The laboratory instrument response versus the unleaded gasoline concentration is shown in Fig. 4. To obtain the data plotted in Fig. 4, serial dilutions were made of stock gasoline solutions in phosphate-buffered water (pH 7.2). A 25-m sensor was used to record the fluorescence intensity for each solution concentration. The highest intensity was assigned a value of 1.0, while the other intensities were normalized to this number. Beer's law predicts an exponential relationship between the intensity and the log of concentration. The line connecting the data points appears to follow an exponential curve. These data show useful laboratory measurements in the parts-per-trillion range, while field results typically show more scatter, with a useful concentration threshold of detection of about 1 ppb. The RLIF response is to the benzene, ethylbenzene, toluene, and xylenes (BTEX) component of gasoline. More complex aromatic molecules, such as humic acid, tend to fluoresce at longer wavelengths than do the single-ring aromatics. Possible interferences from humic acid can be eliminated by use of glass cutoff filters in the detector module [1]. In this approach, such cutoff filters are placed in the light path before the detection PMT. They screen out the fluorescence from humics, while passing the light from BTEX fluorescence into the detection PMT.

Discussion

Since it measures fluorescence, RLIF is a technique that offers speed, sensitivity, and low cost. Its main defect at this time is its lack of specificity. If a mixture of aromatics is present, the RLIF result will be the total signal from *all* fluorescent species present. Keeping this in mind, there are still many potential uses for this technique. If used in conjunction with other, more sophisticated, more time-consuming, and more expensive methods, useful information about changes in an aquifer system can be obtained in a timely and cost-effective manner. For example, RLIF screening could determine if aromatics are present in a well. The exact makeup of a mixture of aromatics could be found using GC or GC/mass spectroscopy (GC/MS). If, then, a site is characterized as having aromatic organ-

ics as the contaminants of concern, the speed and ease of RLIF measurement makes it an attractive alternative to long-term monitoring for changes in the contaminants present. Currently, the limits of detection for RLIF are at or below those for GC analysis, so that the sensitivity of analysis will not suffer.

Limited data from field tests show that the field performance of RLIF is poorer than the laboratory results. Temperature fluctuations in the field, as well as increased vibration, affecting the optics, may be contributing to the variability in field measurements. This problem needs further work.

If a mixture of aromatics is present, the RLIF signal will be a combination of those from each component. If information about each component is needed, separation of the contribution from each component in the mixture would require mathematical deconvolution of the total signal. In baseline monitoring, however, a simple change in the total signal would indicate a change in aromatics concentration in the ground water. Such a simple, inexpensive, and rapid test for change in aromatics concentration in an aquifer would not require extensive deconvolution of the data obtained. Merely following the trend in RLIF results for a given site over time should indicate the presence or absence of a problem involving changing concentrations of aromatic organics, which still may be confirmed by annual GC measurements. The authors hope that the advantages of *in situ* RLIF monitoring will make it an attractive alternative to current methods.

Conclusions

The use of laser-induced fluorescence and fiber optics to measure aromatics contamination of ground water is providing a new tool to the monitoring community. Future pilot and field testing of the current prototype should show the relative advantages of RLIF over traditional methods.

Acknowledgments

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References

- [1] Chudyk, W. A., Kenny, J. E., Jarvis, G. B., and Pohlig, K. O., *InTech*, Vol. 34, No. 5, 1987, p. 53.
- [2] Kenny, J. E., Jarvis, G. B., Chudyk, W. A., and Pohlig, K. O., "Remote Laser-Induced Fluorescence Monitoring of Groundwater Contaminants: Prototype Field Instrument," *Analytical Instrumentation*, Vol. 16, No. 4, 1987, pp. 423-446.
- [3] Eastwood, D., in *Modern Fluorescence Spectroscopy*, Vol. 4, E. L. Wehrey, Ed., Plenum Publishing, New York, 1981, pp. 251-275.
- [4] Eccles, L. A., Simon, S. J., and Klainer, S. M., "In Situ Monitoring at Superfund Sites with Fiber Optics (Draft)," U.S. Environmental Protection Agency Environmental Systems Monitoring Laboratory, Las Vegas, NV, March 1987.

Monitoring in Karst

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Special Problems of Ground-Water Monitoring in Karst Terranes

REFERENCE: Quinlan, J. F., "Special Problems of Ground-Water Monitoring in Karst Terranes," *Ground Water and Vadose Zone Monitoring, ASTM STP 1053*, D. M. Nielsen and A. I. Johnson, Eds., American Society for Testing and Materials, Philadelphia, 1990, pp. 275–304.

ABSTRACT: Reliable monitoring of ground-water quality in any terrane is difficult. There are many ways in which violation of sound principles of monitoring-network design and good sampling protocol makes it easy to acquire data that are not representative of the water or pollutants within an aquifer. In karst terranes it is especially easy for irrelevant data, which inadvertently misrepresent conditions within the aquifer, to be obtained.

The special problems of monitoring ground water in most karst terranes can be grouped into four major categories of problems that are rarely as significant in other terranes. These categories are the following:

1. *Where to monitor for pollutants:* The only relevant locations are at springs, cave streams, and wells that have been shown by tracing tests to include drainage from the facility to be monitored—rather than at wells to which traces have not been run but which were selected because of their convenient downgradient locations. Wells located on fracture traces and fracture-trace intersections and wells located randomly can be successfully used for monitoring, but only if there is a positive trace from the facility to them. Often successful monitoring can only be done several kilometres away from the facility.

2. *Where to monitor for background:* The only relevant locations are at springs, cave streams, and wells in fractured rock—in which the waters are geochemically similar to those to be monitored for pollutants but which have been shown by tracing tests *not* to drain from the facility—rather than at wells selected because of their convenient locations upgradient from the facility site. This, too, may have to be done several kilometres away from the facility.

3. *When to monitor:* Before, during, and after storms or meltwater events—rather than regularly with weekly, monthly, quarterly, semiannual, or annual frequency.

4. *How to determine reliably and economically the answers to Problems 1, 2, and 3:* Reliable monitoring of ground water in karst terranes can be done, but it is not cheap or easy.

These problems exist because many of the assumptions made for monitoring flow in granular media are not valid for karst terranes. Implicit assumptions made for monitoring in karst terranes with the strategy recommended herein can be stated axiomatically, but they are valid only about 95% of the time.

The monitoring strategy recommended herein is not applicable universally, but it is applicable in most karst aquifers, especially, all those that drain to springs. It is not applicable in terranes that are merely recharge areas of regional aquifers.

KEY WORDS: ground water, ground-water monitoring, karst, water quality, springs, hazardous wastes, limestone, carbonate rocks

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Approximately 20% of the United States consists of areas underlain by limestone or dolomite; most of these areas are karst terranes. Sinkholes, sinking streams, losing streams, gaining streams, springs, and caves commonly occur, but they may be absent or not obvious. A karst terrane generally includes a karst aquifer, an aquifer occurring chiefly in limestone, dolomite, gypsum, salt, carbonate-cemented clastic rock, or some combination of these rocks. Shale, siltstone, or sandstone may be interbedded in a karst aquifer. The most characteristic feature of a karst aquifer is flow of water through conduits (caves) and along bedding planes and fractures enlarged by solution.

Waste-disposal facilities should not be located within a karst terrane unless those involved are willing to risk sacrificing the use of at least part of the included karst aquifer as a source of potable water. This is a high risk, almost a certainty. Nevertheless, many facilities already exist within karst environments, and hazardous materials are disposed of without the supervision of a hydrologist knowledgeable about karst. Also, hydrologists are not consulted on the selection of the site when there is an accidental spill of hazardous material.

Two major types of flow occur in karst aquifers—conduit flow and diffuse flow, each of which is an end member of a continuum. Springs and cave streams in conduit-flow systems are “flashy”; that is, they have a high ratio between the maximum discharge and the base-flow discharge, typically 10:1 to 1000:1. The discharge responds rapidly to rainfall, and the flow is generally turbulent. The waters possess low but highly variable hardness, and the turbidity, discharge, and temperature of these waters also vary widely [1]. Where a karst aquifer is less developed and is characterized primarily by diffuse flow, its behavior is less flashy; the ratio between the maximum discharge and base-flow discharge of major springs is low (4:1 or less), and the response of their discharge to rainfall is slow. Flow is generally laminar. The hardness in such springs is higher than that in conduit-flow springs, but the hardness, turbidity, discharge, and temperature have low variability [1]. The variations in and relationships among these properties and their variability as a function of aquifer flow, storage, and recharge have been described in an important paper by Smart and Hobbs. [2].

The relationships between conduit-flow aquifers, fractured aquifers, granular aquifers, and diffuse-flow aquifers are shown in Fig. 1. Ground-water flow in conduits and fractures of karst aquifers differs radically from flow in other aquifers. Most commonly, flow in karst is to springs, by way of caves. Such flow is generally faster than in other aquifers; extreme velocities of 2300 m/h (7500 ft/h) have been observed, while a range of 10 to 500 m/h (30 to 1500 ft/h) is common between the same two points (Ref 1, page 202). [The latter two flow velocities are equivalent to approximately 90 and 4400 km/year (55 and 2700 mile/year).] Accordingly, the effects of leakage or a spill of hazardous material on the water quality in a karst aquifer can often be sensed long distances away within less than a day.

To many hydrologists, geologists, and engineers, the flow of ground water in karst is mysterious, capricious, and unpredictable. Few publications adequately discuss the predictive aspects of environmental hydrologic problems of karst terranes or offer practical, experience-based insights for solving them. There are, however, notable exceptions [1–13]. The problems of flow prediction are real, yet they are ignored in nearly all of the environmental monitoring literature and by most current U.S. Environmental Protection Agency (EPA) and state ground-water monitoring regulations and technical enforcement guidance documents. These problems, nevertheless, can be solved. Education and awareness of their existence are the keys for doing so.

This paper discusses where and when to take relevant samples and how to get the data essential for making the where and when decisions; it also discusses implicit assumptions made when monitoring with the strategy recommended herein. No attempt is made to discuss the design or construction of monitoring wells, protocol for sample custody, or

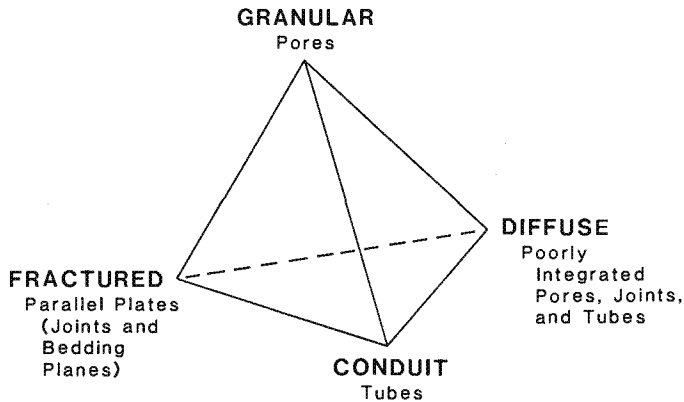


FIG. 1—Relationship between the four major types of aquifers and the dominant porosity geometry in each. This is a tetrahedral continuum.

quality assurance/quality control (QA/QC). These topics have been adequately addressed by others, and many are being reviewed and codified by members of ASTM Subcommittee D18.21 on Ground Water Monitoring and Vadose Zone Investigations, a subcommittee of ASTM Committee D-18 on Soil and Rock. This paper is prescriptive and practical rather than philosophical. The methodologies for its recommendations and the rationale for each are discussed in the references cited. Although the various answers to where, when, and how to monitor are strongly influenced by the answers to *why*, this paper does not address the complex questions of why to monitor.

The recommended strategy for traced-spring, traced-cave-stream, and traced-well monitoring, described herein and by Quinlan and Ewers [1], is not applicable universally. But it is applicable in all diffuse-flow and conduit-flow aquifers that drain to springs which discharge on land or along the shores of streams, rivers, lakes, or seas. Accordingly, the strategy is applicable in most karsts in the following 16 states, which have significant amounts of karst: New York, Pennsylvania, Maryland, West Virginia, Virginia, Tennessee, Georgia (Appalachians), Alabama, Kentucky, Indiana, Arkansas, Missouri, Iowa, Minnesota, Texas, and Oklahoma. Their karst terranes are characterized by local recharge *and* discharge. Many of these states include some of the more densely industrialized and populated areas of the United States. Karst comprises approximately 25 to 30% of the total area of these states.

The recommended monitoring strategy is applicable only locally in parts of the Floridan aquifer of Florida and in Puerto Rico. In both areas there is significant discharge at springs. The strategy is not applicable in karst terranes that are merely recharge areas of regional aquifers, such as the Upper Floridan aquifer of Florida, Georgia, and South Carolina. It is also minimally applicable in karsts mantled by glacial sediments and in which the discharge is diffuse, flowing into sediment or bodies of water rather than to discrete springs. These karsts comprise an estimated 30% of the total area of these states.

The strategy would be applicable in most of the Edwards aquifer of Texas, much of the Upper Floridan aquifer, and part of the North Coast limestone aquifer of Puerto Rico, where the flow is to springs at the surface and where most springs are diffuse-flow rather than conduit-flow springs. Although much of each of these aquifers is characterized by sponge-like permeability, many of their springs are fed by conduits that are commonly braided (anastomosed). Accordingly, one might use geophysical techniques for trying to

find the main conduit and drill to intercept it, but probably would miss it. Nevertheless, monitoring could be successfully accomplished at springs. Wells in these sponge-like aquifers could be used as monitoring sites only if there were a positive trace to them from the vicinity of a facility or from the facility itself during low-, moderate-, and high-flow conditions.

There are numerous small areas of karst in the western United States, but nearly all of them are in isolated, nonindustrialized, unpopulated terranes.

Research is needed on the distribution of and criteria for applicability of the recommended strategy in karsts in the states cited in the four preceding paragraphs.

Ignored here is the fact that most current regulations concerning ground-water monitoring disregard the manifold problems of doing it reliably in karst terranes. The true goal of ground-water monitoring should be to detect the nature and magnitude of changes, if any, in ground-water quality as a result of natural processes or human activity—rather than just to comply with the letter of the law and local regulations. One assumes that federal and state regulations will eventually catch up with reality and will ultimately reflect an emphasis on the *spirit* and *intent* of the law, rather than mere compliance with its letter.

Design of Monitoring Systems in Karst Terranes

Where to Monitor for Pollutants

Conventional minimum monitoring protocol, as required by most regulatory agencies, begins with drilling of a “sufficient” number of wells. Usually this number is a minimum of four—one well upgradient from a facility and three wells downgradient from it, commonly near the property boundary and almost always on the property. Sampling is performed annually, semiannually, and perhaps quarterly, but rarely as often as monthly. Such monitoring wells in a karst terrane may generate great amounts of carefully collected, expensive data, most of which may be of little value because the wells fail to detect the contaminants they are intended to intercept. They fail to do so because the wells do not encounter the flow lines (cave streams) draining from the site, or—if they do intercept them—an insufficient number of samples is taken or sampling is done at inappropriate intervals and at the wrong time [1,3].

The easiest reliable sites at which to monitor ground-water quality in a karst terrane are springs and directly accessible cave streams, shown by dye tracing to drain from the facility being evaluated [1]. Tracing in karsts is usually done with fluorescent dyes because they are the most cost-efficient of many possible tracers. However, use of the term *dye tracing* in this paper is not intended to exclude the use of other tracers.

The preferable alternative for monitoring sites in karst terranes is a suite of wells that intercept cave streams known (by tracing) to flow from the facility to the spring (or springs) that drain the ground-water basin in which it lies. Cave streams may be difficult or impossible to find with traditional geophysical techniques. New geophysical techniques include the use of streaming potential (measurement of the electrical potential gradient caused by displacement of ions from fissures and rock grains as water moves from a recharge area to a discharge area) [14] and the use of acoustic detection (measurement of sound waves caused by the knocking of pebbles against one another during saltation, by cavitating water, or by cascading, riffing, and dripping water in partly air-filled cavities). These methods have been tested with encouraging results [14]. In one situation, however, the results were not supported by the data presented [15]. Other nontraditional geophysical techniques also have promise [14].

A second alternative for monitoring sites is a suite of wells located in fractured rocks,

preferably on fracture traces or at fracture-trace intersections. These wells are usable *only* if tracer tests show a connection with the facility under base-flow as well as flood-flow conditions. Randomly located wells could also be used if, again, tracing has first proven a connection from the facility to each of the wells under various flow conditions.

Although some cave passages are coincident with various types of fracture traces and lineaments, not all coincident fracture-related features are vertical and therefore directly above cave passages [16]. More important, many cave streams are developed along bedding planes and are unaffected by vertical fractures. This fact lessens the probability that a well drilled on a fracture trace, a lineament, or the intersection of such linear features will intercept a cave stream. This fact does not challenge the well-known correlation of such linear features with increased water yields [17].

Success in the use of tracing to determine those randomly located wells (domestic, agricultural, or industrial) that intercept flow from a site can be maximized if they are pumped continually to discharge at a low rate, say 4 L/min (1 gal/min), through a passive dye-detector (explained here) which is regularly changed once or twice a week for weeks or months. In most settings this small amount of water can be wasted onto the ground at a reasonable distance from any building or structure with no adverse effect. Pumpage at high rates, say 400 L/min (100 gal/min) or more, may distort the flow field near a well, which is acceptable if the effects of such distortion are recognized. A high pumpage rate for weeks to months is expensive and wasteful; furthermore, disposal of the pumped water can be a problem, especially if it is contaminated by pollutants.

If a well is drilled specifically for monitoring of pollutants, one might argue that it would be more useful if it were to be pumped intermittently on a regular schedule, even when it is not actively being used for a dye trace. This intermittent, nonrandom pumping would allow the monitoring well to simulate more truly the flow conditions around a regularly used well that might be contaminated.

At most locations in karst terranes, proper and reliable monitoring (using springs, cave streams, and wells traced positively) can only be done off the facility. Sometimes this proper monitoring must be, and can only be, done at sites up to several kilometres away from the facility.

Where to Monitor for Background

If randomly located, nontraced wells are irrelevant for monitoring possible contaminants in karst ground water, they are equally irrelevant for monitoring background for contaminants. Therefore, springs, cave streams, and wells in settings geochemically similar to the traced monitoring sites are the only suitable places to monitor background meaningfully. This is true, however, only when these places have been shown by carefully designed, repeated dye traces, done under conditions ranging from base flow and flood flow, *not* to drain from the facility. One must be exceedingly cautious in interpreting negative results of a tracer test [18]. Some randomly located wells that are negative in a judiciously designed, properly executed dye trace are in settings geochemically similar to those of the monitoring wells that intercept cave streams; most are not.

Background data at a new facility can also be obtained from wells that have been monitored before, during, and after storms for at least a year before operations begin. Analysis of continuous records of the water-level stage in these wells can be used for selection of storm-related sampling frequencies and for possible differentiation of wells into several categories. Traces should be run during that year-long period in order to demonstrate the presence or absence of connections between the various wells and the facility to be monitored. If, after allowance is made for flow velocity, a well does not test positive for dye, the

well does not comprise an effective monitoring system. If none of the wells, or an insufficient number of wells, to which dye traces have been attempted test positive, an *effective* monitoring system does not exist; probably, the hydrology of the facility was not adequately understood. More wells may have to be drilled and tested by tracing.

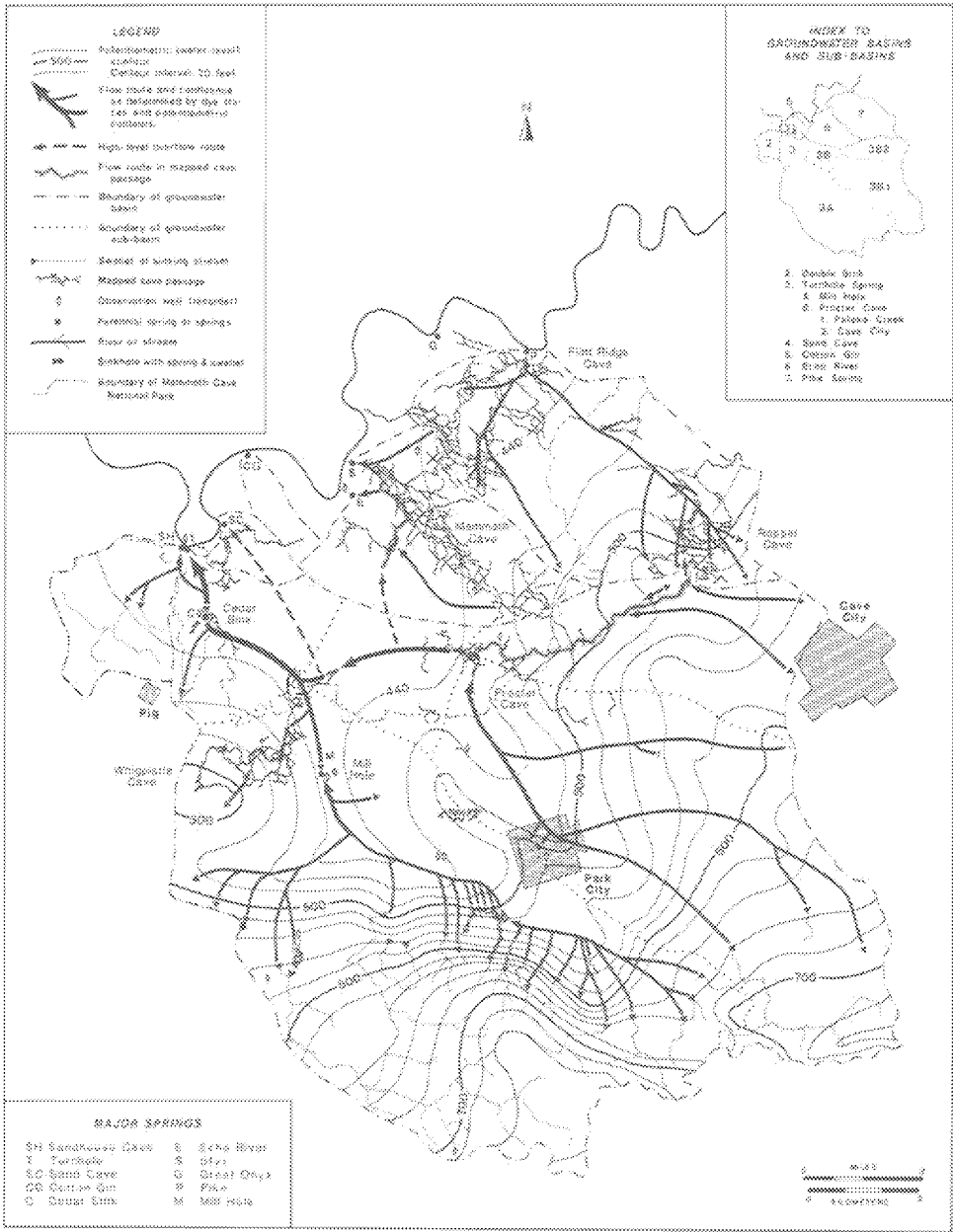
It is tempting and all too easy to take false comfort in the interpretation of negative tracing results for a well to mean that a waste disposal facility and its liner (if present) are functioning as designed. More specifically, it is easy to say that a facility is either not leaking or is adequately attenuating everything put into it, but it would be naive to do so. Alternatively, the negative tracing results could just as logically (indeed, more probably) be a consequence of monitoring for dye in wells that do not sense the part of the aquifer in which ground water actively circulates. In most karst terranes, the latter explanation is more likely to be correct. If water is not standing on the ground in pools or ponds, rainfall must be flowing someplace. If it is flowing, an uncapped facility is probably leaking to somewhere.

There are statistical problems associated with the validity of comparing data from background wells with purportedly relevant data from monitoring wells. Some regional offices of the EPA, in recognition of these problems, are looking for ways to make objective, legitimate comparisons. One way they are using is employment of the concept of "trigger levels" for determining whether or not a facility is in compliance with regulations. Nationwide trigger levels for a compound are being set after critical review of data on toxic effects of that compound on health. If the trigger level of that compound is exceeded in a monitoring well for a facility, an investigation of possible sources for it must be made. No matter what concept of background is used, and no matter what the concept of background evolves into, there is still the problem, addressed in this paper, of how to compare legitimately relevant water-quality data.

Where to Monitor for Pollutants and Background: Other Aspects

All six types of monitoring sites and background sites—springs, directly accessible cave streams, wells drilled to cave streams, wells drilled on fracture traces, wells drilled on fracture-trace intersections, and wells drilled randomly—must be tested by tracing, not only during moderate flow but also during flood flow and base flow, in order to prove the usefulness of these sites for monitoring. This must be done during the extremes of expected flow conditions; flow routing in karst terranes commonly varies as the flow stage changes. During flood conditions the water in some conduits may rise, temporarily move through conduits that are dry during low-flow conditions, and be "decanted" (switched) into adjacent ground-water basins, thus being temporarily pirated by them. An example of such hydraulic switching is depicted in Fig. 2, which shows the hydrology of Turnhole Spring in Mammoth Cave National Park, Kentucky. During moderate flow and flood flow, water draining from Park City and from the west boundary of Cave City in the Turnhole Spring ground-water basin of Kentucky may flow via intermediate-level and high-level crossover routes, shown by the north-trending dashed lines with arrows, to as many as three other basins: No. 4, Sand Cave; No. 6, Echo River; and No. 7, Pike Spring (indicated at the south end of Roppel Cave). These three subsurface diversion routes are also shown in Fig. 3, but more schematically. Other, more complex examples of hydraulic switching are known [19–22].

Another peculiarity of water movement in many karst aquifers is distributary flow. An *underground distributary* is a dispersal route analogous to the distributary at the mouth of a major river, but its origin is different (Ref 1, pages 205 and 207). Figure 3 shows numerous distributaries. The positions and geometry of the underground bifurcations shown are



schematic, but their existence in the map area is a certainty. Some of these underground flow paths have been mapped. Knowledge of the occurrence and functioning of distributaries is important in the design of a monitoring system in a karst terrane because pollutants from a point source in the headwaters or midreaches of a ground-water basin or sub-basin will flow at higher stages to all springs in its distributary system or subsystem. For example, pollutants reaching ground water at a point source in the east-central area of Fig. 2, south of Green River, would—depending upon the flow conditions—disperse to a total of as many as 52 springs in 19 isolated segments along a 19-km (12-mile) reach of the Green River. Not all springs between the extremities of a distributary system are necessarily a part of the system.

The extent to which ground-water basins and their subsurface flow routings can be deciphered by tracing, mapping of the potentiometric surface (water table), and mapping of caves is shown in Fig. 4. The east half of this map summarizes some of the results of a study of the dispersal of heavy metals from a metal-plating plant; the metals had been discharged into the ground (in concentrations of more than 10 mg/L) via a municipal sewage treatment plant at the town of Horse Cave. (The outline of this town is just east of the center of the map.) A study of the chemistry of springs, wells, cave streams, and the west-flowing Green River showed that effluent from the sewage treatment plant flows underground 1.6 km (1 mile) northeast to the unmapped cave beneath the town and then, depending upon the flow stage, to a total of as many as 46 springs in 16 segments along an 8-km (5-mile) reach of the river, about 7 km (4 miles) to the north [23]. None of the 23 wells monitored for heavy metals during base flow showed concentrations higher than background levels, but they should have also been sampled during flood-flow conditions, when water movement might have been reversed from the trunk conduit to some of the wells.

Figure 5 shows complex, radial flow in faulted, flat-lying rocks in the Ozarks of Arkansas. The divergent results of six dye tests, three of which were performed within a mile of a proposed landfill, are summarized. Although the flow routing in Fig. 5 is more complex than that shown in Figs. 2 and 4, it is totally in agreement with the advocated maxim, "Monitor the springs!" Monitoring wells that could be drilled on fracture traces at the proposed landfill site might detect seepage of leachate from it, but there is no means—other than by tracing—to identify correctly and conclusively the places to which leachate would (and probably would not) flow. Stated another way, no matter how superbly efficiently the hypothetical monitoring wells on the landfill property might be able to detect leakage from the landfill, there is no way—other than by tracing or by monitoring of numerous springs and wells off the property—that one could discover the many consequences of leakage from the proposed landfill. This statement would be true even if the monitoring wells were properly constructed and if they functioned efficiently.

All four wells at the town of Pindall, Arkansas (Fig. 5), which were pumped continuously during the dye trace from the east boundary of the landfill site (the first trace run), tested positive for dye. By inference, if other wells in the town had been pumped continuously, many of them, perhaps all, would have tested positive. Note that none of the four wells immediately east and southeast of the first dye-injection site and neither of the two wells immediately west of the western dye-injection site tested positive for dye during the first test. (None of these six wells was sampled for dye during the five subsequent tests.) During the second and third dye tests, in which dye was injected 1.6 km (1 mile) east and west of the proposed landfill site, respectively, only two wells at Pindall were pumped continuously. The southernmost well there tested positive for dye in both tests, as shown; the easternmost well was ambiguous for dye from the east and negative for dye from the west. The high-yield well on the fracture trace tested positive for dye from both the second and

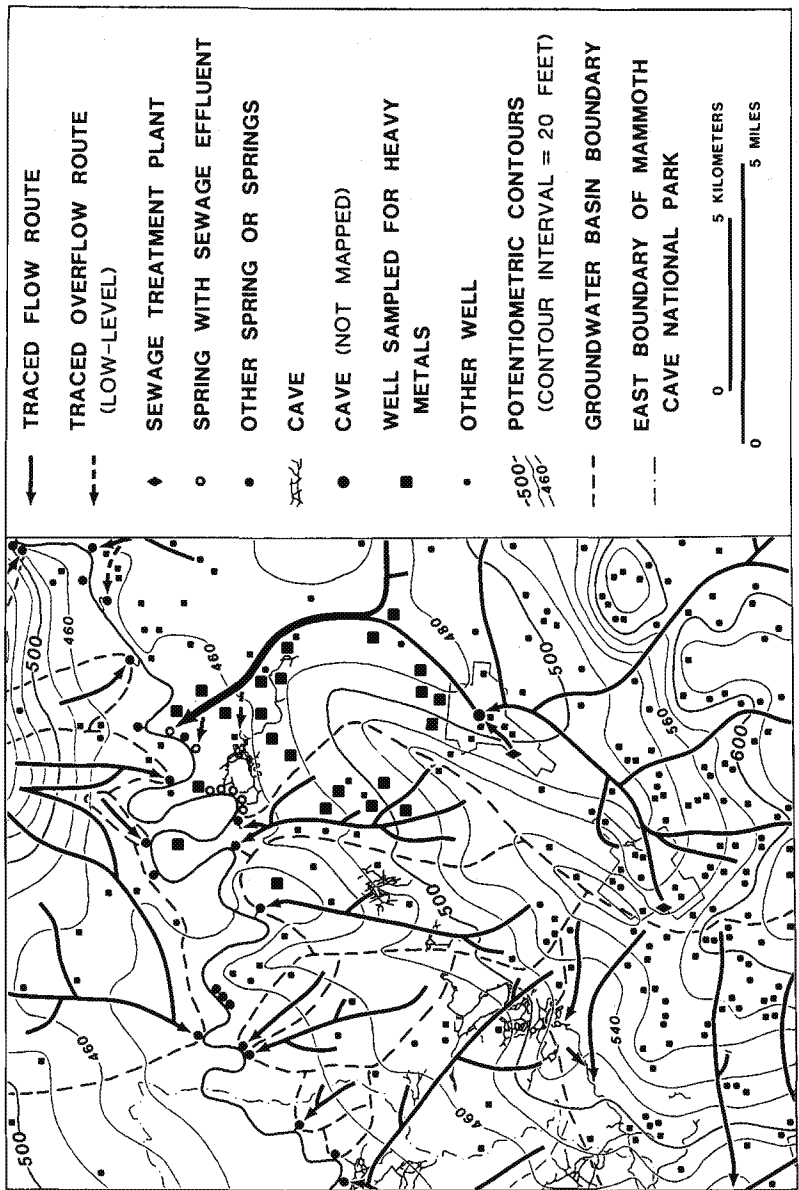


FIG. 4—Ground-water flow routes and pollutant dispersal in the vicinity of Horse Cave and Cave City, Kentucky. Part of the east half of Fig. 2 overlaps with the west half of this map (modified from Ref 12).

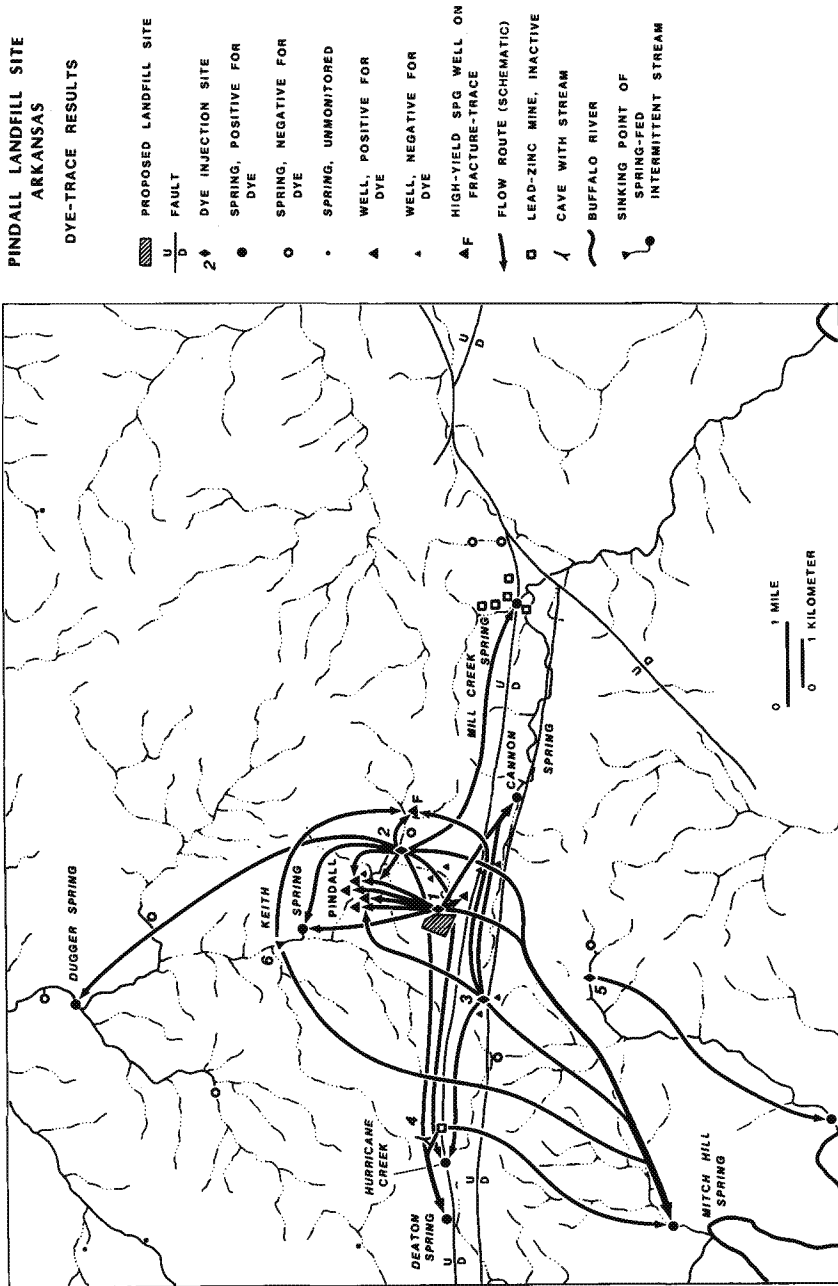


FIG. 5—Complex radial flow of ground water near a proposed landfill in flat-lying, residuum-mantled limestone in the Ozarks of Arkansas. These results are based on six dye traces, all of which were done near the headwaters of several surface streams and at low-to moderate-flow conditions of ground water, but neither during base flow nor at flood flow. An upper and lower karst aquifer, faulting, fracture traces, lineaments, spillover levels, differences in sampling and analytical protocols, and the intrinsic complexity of flow within the karst all influence the divergent results of the traces. The results might be different if the dye tests had been run during base-flow or flood-flow conditions [24].

third dye tests, but negative for dye from the first. Dye recovery from this well during the first test was probably hampered by chlorine added to the well water, which tends to react with and destroy low concentrations of any dye present. The difference in tracer results at this high-yield well may also have been a consequence of greater efficiency in sampling and analysis during these latter tests and of the use of the author's pumped-well dye-sampling device [24]. (Dye trace results from the first test were instrumental in the 1987 defeat of a proposal to use this site as a landfill.)

Radial flow occurs in many karst terranes and has also been documented at waste disposal facilities (Ref 1, pages 214–219); it tends to be associated with locations on topographic highs. When evaluating a facility by dye tracing, one must keep an open mind and place detectors for dye at not only the logical, obvious places, but also at the illogical, the unlikely, and the “no, it could never go there” places. The cliché “expect the unexpected” applies, no matter how experienced one is in tracing.

The applicability of the spring and cave-stream monitoring strategy and the conventional (randomly located well) monitoring strategy in various types of aquifers is shown in Fig. 6.

The monitoring strategy advocated here and discussed in more detail elsewhere [1,25] works. It is a significant advance over traditional monitoring strategies and has been recognized as such [26], but it is far from the last word on the subject. Much remains to be learned and documented.

When to Monitor for Pollutants and Background

Current conventional monitoring protocol generally requires sampling wells annually, semiannually, occasionally quarterly, and rarely as often as monthly. Such practices are prudent in most nonkarst terranes. In karst terranes, however, even at springs and cave streams judiciously and correctly selected as monitoring sites by the dye-tracing procedures recommended herein and elsewhere [1,25,27], the analytical results of data collected at such regular intervals can be inadvertently misleading, and the net result is a waste of time and money. Why is this? Because, if a karst aquifer is characterized by conduit flow, the chemical quality of the water at a spring to which it drains can be greatly affected by the effects of storms and meltwater events [3,28,29]. Diffuse-flow systems are generally only slightly affected by precipitation events [1].

In order to characterize the natural, storm-related water-quality variability of a spring in a conduit-flow system reliably, sampling must be done much more frequently than has been customary in the past. An attempt to characterize water quality when using typical semiannual sampling for study of a conduit-flow system is analogous to estimating annual rainfall of an area solely on the basis of rainfall data collected on the same two days of each year.

The effect of sampling frequency on the accuracy of characterizing and depicting this storm-related variability in water quality is seen in Fig. 7. Figure 7 is a composite of three published figures [28]. Discharge was recorded continuously. Pesticides were sampled up to 6 times per day, nitrate was sampled up to 20 times per day, and suspended sediment was sampled up to 17 times per day, depending upon the flow stage [30]. For the sake of discussion, however, let us *assume* that these parameters were monitored continuously for the 11-day interval shown. (Assume also that the apparent variations in water quality as a result of its natural variability, the statistics of sampling and analysis, and the analytical error are trivial—even though they possibly are not.) The most important thing shown is that pesticides and suspended sediment have a maximum level that is approximately coincident with that of the discharge. Nitrate, however, has its minimum when the others are

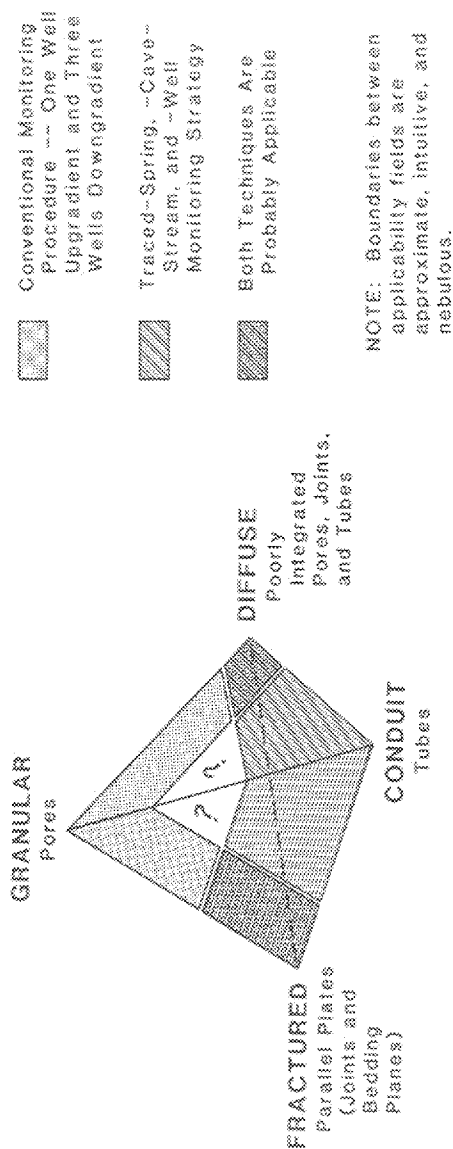


FIG. 6—Diagram showing a tetrahedral continuum between the four major types of aquifers, the dominant porosity geometry in each type of aquifer, the applicability of the spring and cave-stream monitoring strategy in each type of aquifer, and the applicability of the traditional monitoring technique (using randomly located wells) in each type of aquifer [51].

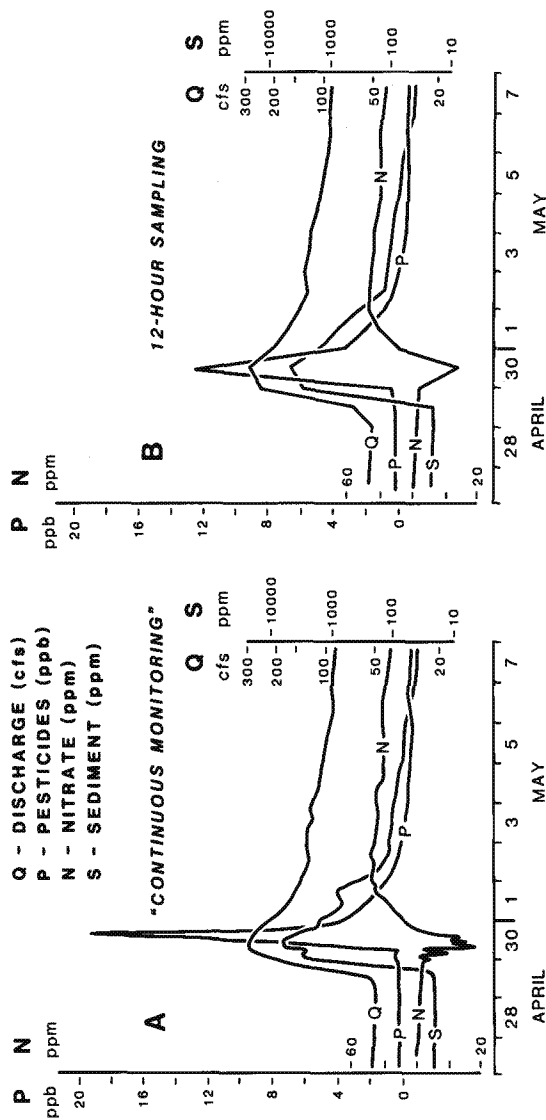


FIG. 7.—Water quality and discharge during a 1984 storm in a karst aquifer characterized by conduit flow, in the Big Spring ground-water basin, Iowa (Q = discharge, in cfs; P = pesticides, in ppb; N = nitrates, in ppm; and S = sediment, in ppm) (modified from Ref 28); (a) continuous monitoring data; (b) 12-h sampling data.

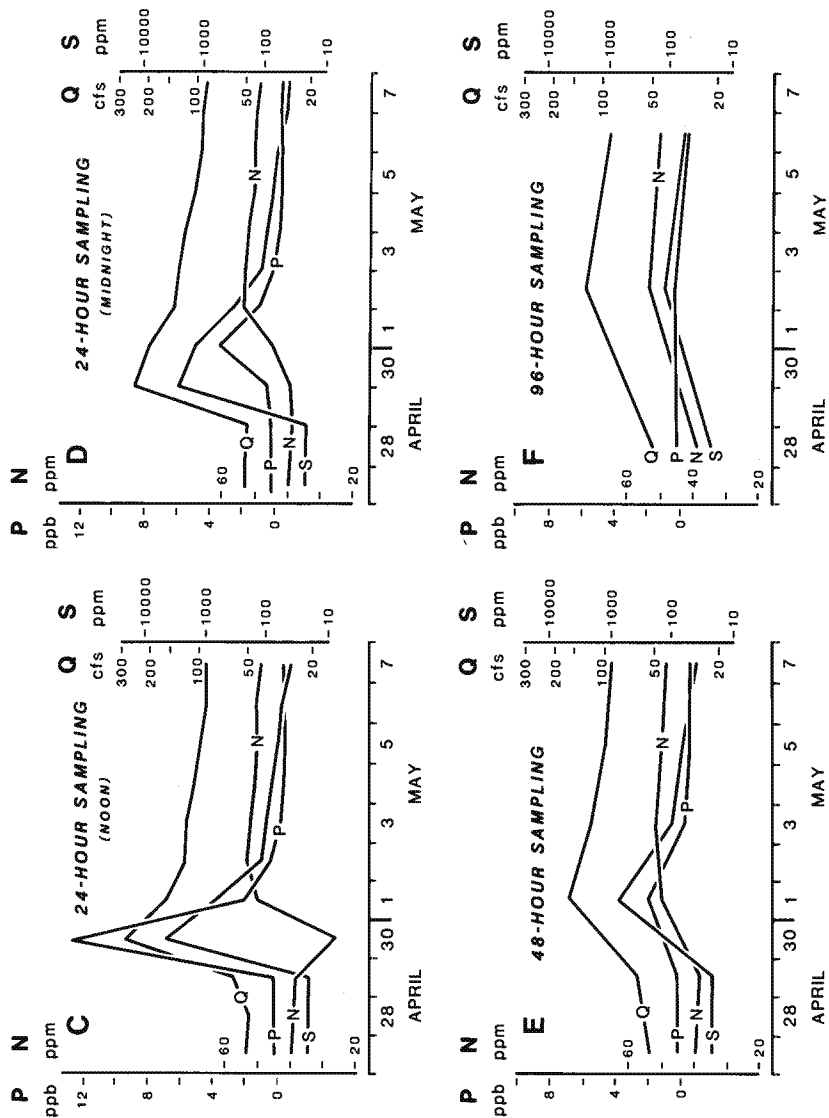


FIG. 7—Continued: (c) 24-h sampling data (noon); (d) 24-h sampling data (midnight); (e) 48-h sampling data; (f) 96-h sampling data.

at a maximum, and it reaches its own maximum several days later when the other pollutants are approaching their low prestorm values. The reasons for this peculiar lack of synchrony are discussed elsewhere [3,28,29]. Note also that pesticide concentrations increased by more than an order of magnitude during and after the storm. Such storm-related variation is common, but longer term, precipitation-related variation in what can falsely be called "background" for pesticide concentration at this site may range over more than three orders of magnitude in just a few months [3]. Such variation in background probably occurs elsewhere.

Figure 7, *b* through *f*, is based on the data represented in Fig. 7*a*, but this figure assumes sampling at intervals of 12, 24, 48, and 96 h. For the storm event that occurred, sampling at 48 and 96-h intervals (Fig. 7, *e* and *f*) is totally incapable of suggesting any significant change in the water quality. The 24-h sampling frequency (Fig. 7, *c* and *d*) is better, but the midnight samples happen to miss the decrease in nitrate and most of the increase in pesticides. Even a 12-h sampling interval (Fig. 7*b*) is only a crude approximation of the continuous sampling represented by Fig. 7*a*.

The author stresses that the "proper sampling frequency," however defined and however determined, will vary with the event to be monitored and with the flow dynamics and flow type (conduit flow versus diffuse flow) of the karst aquifer studied. For example, the sampling frequency necessary to characterize changes accurately in the chemical composition of spring discharge in the Big Spring ground-water basin (as in many other conduit-flow basins in the Mammoth Cave area and elsewhere) will, for similar storms and similar antecedent moisture conditions, be far more than is necessary to provide the same accuracy of characterization at the diffuse-flow springs draining the Edwards aquifer in Texas, the Floridan aquifer, and much of the Ozarks of Missouri and Arkansas.

In September 1987 the EPA announced proposed standards for the concentration of various pesticides in ground water. For atrazine, a herbicide used to control weeds in corn, sorghum, sugarcane, pineapple, and citrus groves, the maximum allowable concentration is 3 ppb [31]. About 90% of the pesticides found in Big Spring basin are atrazine. Therefore, the peak pesticide concentrations shown in Fig. 7, *a* through *c*, are about four to six times higher than this 3 ppb limit; midnight sampling barely detected this violation of the 3 ppb maximum (Fig. 7*d*), and the 48-h and 96-h sampling (Fig. 7, *e* and *f*) totally failed to do so.

Statistical procedures for designing and evaluating sampling strategies are available [32–39] but they are complex. A reliable, cookbook approach to this problem for karst areas would be desirable.

Until an acceptable, economically realistic, reliable procedure for sampling ground water in karsts is developed and tested, probably the best protocol is that proposed and discussed in Ref 3 (page 281). In brief, sampling should start at base flow, before the beginning of a storm or meltwater event, and continue until 4 to 30 times the time to the hydrograph peak has elapsed, depending upon the extent to which an aquifer is characterized by conduit flow as opposed to diffuse flow. Sampling may have to be done as often as at 1 to 6-h intervals in the early part of a precipitation event and at 4 to 24-h intervals in the waning part of its hydrograph.

After a precipitation event, the decision about which samples to analyze, if any, should be based on a careful evaluation of the significance of the event, interpretation of the flow stage and conductivity data from it and previous events, and an estimation of the data needed to characterize the monitoring site. Many samples, sometimes all, can be rightfully discarded. Difficult, judicious decisions must be made. These analytical data must be compared with those for samples taken several times per year, during base flow, storms, and meltwater events. Only then can one possibly make a reliable assessment or characteriza-

tion of the true quality of water draining from a facility. After several years of data have been accumulated and the aquifer behavior is understood, sampling frequency may be decreased.

On the basis of conceptual models, experience, and limited data, the author believes that monitoring in karsts characterized by diffuse flow is easier and cheaper because far fewer samples are required. Also, the more a karst aquifer is characterized by diffuse flow, the higher the probability that fracture-trace-sited wells and randomly located monitoring wells can be used reliably. All wells proposed for monitoring use, however, must still test positive by dye tracing.

A quick, inexpensive way to distinguish between a conduit-flow spring and a diffuse-flow spring is to observe its turbidity (or lack of turbidity) and to measure its specific conductivity before, during, and after several large storms. If the spring is characterized by conduit flow, it will be turbid; the coefficient of variation (standard deviation \div mean \times 100) of its specific conductivity will be from 10% to more than 25%. If the spring is characterized by diffuse flow, its water will always be clear to slightly turbid; the coefficient of variation of its specific conductivity will be less than 5% [1,25]. Remember, however, that conduit flow and diffuse flow are end members of a continuum.

Much remains to be learned about when to sample ground water in karst terranes. The likely possibility of either deliberate or inadvertent acquisition of falsely negative data from these terranes makes it imperative that people in charge of sampling and officials in charge of evaluation of sample data have an understanding of karst problems and carefully scrutinize all analytical results from such terranes.

How to Determine Where and When to Monitor Ground Water in Karst Terranes Reliably and Economically

Where to monitor is typically best determined with the aid of two types of field investigation: dye tracing and mapping of the potentiometric surface. Geophysical studies, such as mapping of the streaming potential of descending waters and mapping of acoustic emissions from cave streams [14], may follow mapping of troughs on the potentiometric surface and precede drilling of wells planned to intercept cave streams on these troughs, which are to be tested by tracing for a connection to a facility.

Under "ideal" circumstances, one can run the dye tests necessary for the design of a monitoring system from a sinking stream on the facility. Often, however, no stream is available for dye injection. How then can introduction of the dye be done? One can use tank trucks of water and inject dye at (in decreasing order of desirability) the following locations:

- (a) a sinkhole with a hole at its bottom;
- (b) a sinkhole without a hole at its bottom (excavation may reveal a hole that can be used);
- (c) a losing-stream reach with intermittent flow;
- (d) a Class V stormwater drainage well;
- (e) a well drilled on a fracture trace or a fracture-trace intersection;
- (f) an abandoned domestic, agricultural, or industrial well; or
- (g) a well randomly drilled for dye injection.

Wells drilled for dye injection should probably extend about 7 m (25 ft) below the potentiometric surface, if there is one, but this is a site-specific determination. Before going to the trouble and expense of conducting a dye test from injection-site Types *b* and *e* through

g, however, these sites should first be given a percolation test with a tank truck of potable water to see how rapidly they drain. Alternatively, a pumping test can be run. An electric tape or a pressure transducer can be used to determine the rate of water-level decline in a well; plots of such data can be used to select what is probably the most direct and open connection to the conduits that drain an aquifer. If the percolation test of an injection site shows little or no drainage, dye should not be injected into it. The procedures for dye tracing with trucked water are discussed elsewhere (Ref 1, page 222) [27].

It is economically almost impossible to design a dye trace that simulates the conditions beneath a landfill in, for example, a terrane characterized by 20 m (60 ft) of residual soil. Why is this? Because, the residuum is anisotropic. The distribution of macropores, which may make its permeability several orders of magnitude greater than that of the bulk of the residuum, is unpredictable [40]. These larger macropores, through which fluids move most rapidly, may be several centimetres to several metres apart. The problem of intercepting them with a drill hole to be used for dye injection is analogous to the problem of searching for ore bodies with a drill hole; only the size of the targets, and therefore the spacing between holes necessary to achieve the same probability of target interception, is different.

The relative suitability of different drill holes in residuum or bedrock can be evaluated by percolation tests, as discussed above. If the holes are very close to one another it would be prudent to have an electric tape or a pressure transducer in the adjacent holes—just to be sure that the rapid fall of the water level in the tested hole is not a consequence of leakage into one or more of the adjacent holes.

If, instead of using drill holes for dye injection, one were to excavate a 10 by 10-m (30 by 30-ft) pit to the depth of the bottom of a proposed landfill in the example terrane, say 5 m (15 ft), and carefully construct either a simulated compacted or lined bottom of a cell—or even try to make an “undisturbed” bottom—one could not be sure of having simulated or tested the *long-term* permeability conditions at the bottom of the landfill.

Therefore, whether using a sinkhole, a drilled hole, or a pit for dye injection, when designing a dye trace to evaluate a proposed landfill site, one must do the following:

1. Assume that the soil or liner has or will have differential permeability (leakage) that cannot be remedied by economically justifiable construction methods. This assumption is supported by an extensive literature (see Ref 1, page 199). The long-term permeability of any kind of liner may be affected by chemical changes induced by the leachate. The urgent questions about leakage are *when*, *to where*, and *how fast*. (This is consistent with the EPA policy of assuming a worst possible case scenario.)
2. Test for the consequences of the leakage that is certain to occur.

Since leakage is reasonably assumed to be a certainty, the dye test should be designed to maximize the probability of getting the dye through the soil or residuum as rapidly as possible. Few consulting firms or their clients can afford to wait a year or more for test results that, until the dye is recovered, are negative. The argument that a dye trace (assuming it is well designed and properly run) is irrelevant to evaluation of a particular landfill because the dye was not injected at the bottom of the actual landfill and precisely at its location is specious.

It is naive, erroneous, and dangerous to assume that all or most of the springs necessary for tracer tests and ground-water monitoring are shown on U.S. Geological Survey 7.5-min topographic quadrangles. The author's experience in numerous karst areas has shown that only about 5% of the springs discharging non-isolated *local* flow are shown on maps. Fieldwork for finding springs is mandatory; there is no substitute!

All springs within a radius of perhaps 8 to 25 or more kilometres (5 to 15 or more miles)

from a facility, especially those within $\pm 90^\circ$ of the likely vector of the hydraulic gradient from it, should be found and probably monitored during dye tracing. A prudent designer of a dye trace will, however, at the beginning of an investigation, generally assume the possibility of radial flow and will have 360° of coverage with dye detectors—if only to defend the test design from criticism of inadequacy. The radius of the spring search is determined by evaluating the stratigraphy, structure, and physiography of the terrane and by proposing various tentative hypotheses about possible flow routes and resurgences. These working hypotheses must be tested.

Dye traces to springs at the bottom of sinkholes are especially important for recognizing hitherto-isolated segments of the plumbing system of a karst aquifer. These segments between a facility and the spring to which it drains can be used for monitoring; they offer two advantages over monitoring of springs: less dilution of pollutants or surrogate compounds, and earlier detection of them.

Imagine how extremely different the tracing results in Fig. 5 would appear if the designer of the dye tests shown had followed a “hunch” and monitored only the springs in one particular direction from the landfill rather than in all directions! At the administrative hearing on whether or not to grant a permit for construction of the proposed landfill—held after the dye test results from an area adjacent to the site were available, but before the other five dye tests had been run—the state’s expert witnesses alleged that it was impossible for dye (or pollutants) to flow in opposite directions. They vigorously but erroneously impugned the validity of the dye test. The five subsequent tests resolved all questions about the alleged impossibility of radial flow.

Prediction of flow within a ground-water basin characterized by local flow is usually very much like prediction of flow within a surface-water basin. It moves “downhill” (downgradient) to the trunk drain at the local base level. If the boundaries of either basin are known, it can be confidently stated that although small-scale local flow from an area (or facility) may be in almost any direction, larger-scale local flow will be to tributaries and ultimately to the trunk that drains the basin. For example, in Fig. 2, any dye input (or pollutant) injected south of Mill Hole and west of Park City can confidently be predicted to flow to Mill Hole. Similar predictions can be reliably made for other potential facilities elsewhere in the Turnhole Spring ground-water basin and for anywhere in the area shown in Fig. 4.

The monitoring system for a facility and the consequences of leakage from it should be tested by tracing dye movement from the facility itself. This may not be possible. If it is not, traces should then be run from sites adjacent to the facility, preferably from opposite sides of it, and at points lying on a line approximately perpendicular to the suspected flow direction. This increases the probability of discovering whether a facility is near the boundary of a ground-water basin and whether it consistently flows to the same spring (or springs). Such tracer results are relevant to objective evaluation of the facility. This principle for selecting sites for injection of tracers is called hydrologic juxtaposition. Its validity is strained by the tracer results shown in Fig. 5 but not contradicted by them.

To summarize the principle of hydrologic juxtaposition, if the geology of a dye injection site is similar to that of a site immediately adjacent, it is highly probable that tracer results from the two sites will show discharge to the same spring. Obviously, this may not be so in the immediate vicinity of the boundary between two ground-water basins, but it is the reason a second test is recommended for the opposite side of a facility. In fact, predictions based on the results of a tracer test run from each side of a facility (on or off it) should provide greater confidence than those based on only a single tracer test run from the middle of the area.

Ground-water tracing can be done with many different tracers, but in general the cheapest, most efficacious ones are fluorescent dyes such as fluorescein (CI Acid Yellow 73),

Rhodamine WT (CI Acid Red 388), CI Direct Yellow 96, and optical brighteners such as CI Fluorescent Whitening Agents 22 and 28. Brief summaries of practical techniques for dye tracing have been published [1,41,42], but a comprehensive, plain-English guide to the use of dyes as tracers is scheduled for publication during 1990 [27]. This guide will include many practical hints and suggestions that will enhance the rigor of test designs with any kind of tracer and will increase the reliability of test results. A sampling device for efficiently recovering various types of dyes from continuously pumped wells has been invented and tested by the author. A review summarizing the nontoxicity of dyes commonly used for tracing has been made by Smart [43].

Three types of dye tracing can be used for evaluating the suitability of springs, cave streams, and wells for ground-water monitoring; they are as follows:

1. Qualitative tracing, using either of the following:
 - (a) Visual observation of the dye plume. Generally this is wasteful of dye and may cause aesthetic and public-relations problems. Also, there is a great risk of missing the dye pulse when it arrives at the monitoring site, especially at night.
 - (b) Passive detection (with passive detectors consisting of activated charcoal or cotton, depending upon the tracer used) plus either visual observation of dye eluted from the charcoal or ultraviolet observation of the cotton [27,41,42].

Qualitative tracing is sufficient for most dye tests; when it is done with passive detectors, it is generally the most cost-efficient tracing technique.

2. Semiquantitative tracing, using passive detectors and instrumental analysis of the elutant or cotton with a fluorometer or a scanning spectrofluorophotometer [27,44,45]. The many variables associated with changes in spring or stream discharge, with the reaction kinetics of sorption of dye onto passive detectors and with elution of dye from the charcoal, all make it impossible to quantify precisely the varying concentrations of dye that passed any specific monitoring site during a given period of time. Instrumental analysis can identify dye concentrations several orders of magnitude smaller than those detectable visually.

3. Quantitative tracing, using instrumental analysis of water samples (either grab samples or those taken with an automatic sampler) or of water continuously flowing through an instrument, preferably (for either option) with continuous measurement of the discharge [19–21,27,46]. Instrumental analysis makes possible more precise determination of flow velocity, the breakthrough curve characteristic of a tracer's arrival and retardation, and the aquifer dispersivity. It also allows calculation of dye recovery. Many interwell traces are done with this type of quantification; some evaluations of wells as potential sites for monitoring can only be done with interwell traces. For a given trace, quantitative tracing is the most expensive procedure, but it can give answers not available by any other technique [19–21,46].

If quantitative tracing results are needed in the design of a monitoring system—and generally they are not—it is commonly far more cost-efficient to first do a qualitative or semiquantitative study. This eliminates substantial costs of sampling and analysis of numerous sites to which no dye travels.

Each of these three types of dye tracing is sufficient and satisfactory for establishing a hydrologic connection between two points. Semiquantitative and quantitative tracing techniques are more sensitive to detection of small concentrations of dye; for litigation, they are more convincing. Quantitative tracing techniques are the most sensitive for detection of small, temporary changes in dye concentration.

No matter which of the three types of dye tracing is used for an investigation, it is impor-

tant to avoid two of the more common mistakes of people inexperienced in tracing: not sampling at enough sites at which dye could possibly be recovered (generally using passive detectors) and not sampling long enough. If not enough sites are sampled or if sampling is stopped too soon after the first positive results, one would fail to detect dye at the other places to which it also goes (if, indeed, it goes elsewhere) at either the same velocity or a different velocity. Also, one would fail to discover if some of the dye had become stored in the *epikarst* (as discussed subsequently under *Exception 3*) and was being released over a long period of time. Any of these consequences of inadequate sampling for dye would prevent discovery of aquifer properties that adversely influence the adequacy of the design of a monitoring system.

Ideal tracers are conservative. They do not react with soil, bedrock, or ground water and they do not undergo microbial decay. Most real tracers, including dyes, are slightly reactive and may undergo adsorption-desorption and cation-exchange reactions. Organic pollutants may undergo similar reactions, which affect their rate of migration. This rate is predictable and generally correlated with their octanol-water partition coefficients (Ref 47, pages 397–405, and Ref 48); these pollutants may also undergo microbial decay. Depending upon their mobility, pollutants may travel faster or slower than a dye. Although tracing velocities can be used as a reliable guide for prediction of flow velocities of pollutants under similar antecedent moisture conditions, especially in conduit-flow aquifers, velocities of conservative tracers in diffuse-flow aquifers (as well as in granular aquifers) will be significantly higher than those of most pollutants.

In dye tracing, wisdom is knowing what essential questions need to be asked, and asking them; experience is knowing the most expedient, most prudent way to get the answers to these essential questions.

Although a dye test is like the birth of a baby—no matter how many men and women are put on the job, it will take however long is necessary to complete the task—it is also quite different. The birth of a baby can confidently be predicted to occur probably about nine months after conception. A dye test, however may be completed within a few hours or days after injection, but it could as easily take weeks, months, and even years. One must be patient while waiting for tracing results—or risk malpractice litigation and loss of valuable data. It might take just a few weeks to do the dye traces necessary to design and test the monitoring system for a facility, but it is likely to require six to nine months of intermittent, careful tracing. Warning: it could take even longer.

The potentiometric surface of a karst aquifer should be mapped with as much data as possible. For basin analysis, a minimum of 1 well per square kilometre (2.5 wells per square mile) is recommended for most aquifers. A facility analysis could require more than 40 times this well density. A carefully contoured potentiometric map, if based on valid measurements of an adequate number of wells, can be used to accomplish the following:

- (a) predict the flow routes of tracers (or pollutants),
- (b) judiciously select dye injection sites,
- (c) minimize the number of traces necessary for evaluation of a ground-water basin or a facility,
- (d) interpolate flow routes in the areas between dye traces, and
- (e) detect the possible influence of shale beds and other poorly permeable rocks on perching and confinement of water within a karst aquifer.

Water levels for mapping the potentiometric surface can be obtained with a steel tape, an electric tape, or an acoustic well probe [1,41,42], but such mapping must include a QA/QC program using a vertically hanging steel tape as a standard before, after, and preferably

during periods of use—especially if water levels more than 30 m (100 ft) are being measured. Permanent tape stretching of as much as 2% is known to occur in electric tapes employing copper twin-lead wire similar to those used for TV antenna lead-in. Water-level measurements generally should be made during low-flow and base-flow conditions, rather than during the rainy season or after storms, when water levels can locally be significantly higher and the surface can have a different configuration.

Implicit Assumptions of Recommended Monitoring Strategy, with Examples of Exceptions

Ground water has been recognized to circulate in three different types of flow systems: local, intermediate, and regional [49,50], as reviewed in Ref 47 (pages 217–258). Many axiomatic, implicit flow-system assumptions are made by those who may use the traced spring-, cave-stream-, and well-monitoring strategy advocated here and in Refs 1 and 25, but one should realize what they are. Seven of these major implicit assumptions follow:

1. Discharge is concentrated at a point (a spring or group of springs) rather than diffused over a broad area or concentrated along a line (such as a stream).
2. Most flow systems to be monitored in karst are characterized by local flow, in the sense of Toth [49,50].
3. Ground-water velocities in karsts characterized by local flow have the high values already cited herein. The apparent velocities in karsts characterized by intermediate-flow systems and by regional-flow systems tend to be several orders of magnitude slower than those in local-flow systems.
4. A ground-water basin is a discrete entity having a specific, well-defined boundary.
5. Ground-water basins are contiguous.
6. Storm-related diversion of ground water out of a basin, if it occurs, is by means of intermediate- and high-level overflow routes (conduits) leading to adjacent ground-water basins. To restate: storm-related piracy of ground water, if it occurs, is a temporary diversion—from a pirator ground-water basin to the original piratee ground-water basin.
7. Temporary, storm-related diversion of surface water within a karst ground-water basin is not a significant problem because all the water will remain within the basin.

On the basis of more than 30 years of experience the author believes that each of these implicit, axiomatic assumptions is correct about 95% of the time—often enough to be fairly assumed until or unless data imply otherwise, but not so often as to be a certainty. One must always ask, “What are my paradigms and their assumptions?” and critically review the validity of each. Systematic analysis and review of examples of probable exceptions to the above implicit assumptions and others could be the subject of another paper [25], but here only one or two exceptions to some of them are cited and briefly discussed, and the potential relevance of these exceptions to a monitoring program is stated.

Exception 1: Nonpoint Discharge

Although some discharge in the karsted dolomite of the Door Peninsula, Wisconsin, is to springs [51], most of the discharge is through sediment and over a broad area beneath Green Bay and Lake Michigan [52,53]. Discharge from a karst aquifer through sediment over a broad area and along a line occurs in the Cano Tiburones area north and west of Barceloneta, Puerto Rico, and along a line in the valley of the Rio Grande de Manati, south of Barceloneta. These terranes are in Tertiary limestones of the alluvium and paludal sed-

iment mantled shallow aquifer on the north coast; the aquifer seems to be characterized by diffuse flow, but springs and conduit flow are locally important [54]. Monitoring in terranes characterized by areal discharge can best be done by using randomly located wells, perhaps along fracture traces, but these wells might not intercept the relevant flow lines. Monitoring in terranes characterized by seepage along a line is best done by sampling at intervals along the line and upgradient from it. In each hydrologic setting, however, although flow velocities may be very low, tracing must be done if confidence in a monitoring effort there is what is desired. If a well proposed for monitoring in a karst terrane does not have a positive trace to it (or to its site), it isn't a monitoring well.

An example of discharge along a line occurs in the eastern Snake River Plains aquifer, in Idaho [55]. Admittedly, this basalt aquifer is not a karst aquifer, but it was because of the hydrology of such rocks that the term *pseudokarst* was first proposed more than 80 years ago (Ref 56, pages 182–183). Many of the monitoring principles advocated herein are applicable to monitoring in this basalt aquifer and in other highly fractured rocks.

Exception 2: Nonlocal Flow

There are many examples of nonlocal flow that must be monitored in karst aquifers. Some of the better known examples of such karsts are the Edwards aquifer [57,58] and the Floridan aquifer (Ref 47, pages 237–243 and 359–361) [59,60]. Flow velocities in them are likely to be much slower than in most conduit-flow aquifers and more like the low velocities characteristic of most diffuse-flow aquifers. Flow paths are likely to be braided and dispersive rather than convergent; they still, however, flow to springs.

Exception 3: Slow Movement in a Local-Flow System

Immediately beneath the soil profile of most karsts is the *epikarst*, a 3 to 10-m (10 to 30-ft) zone of karstification in which horizontal flow is dominant and storage is significant (Ref 9, pages 28–35) [1,2,61–64]. (Some authors prefer to use the synonymous term *subcutaneous karst*.) Usually the epikarst is separated from the phreatic zone by a dry, inactive waterless interval of bedrock that is locally breached by vertical percolation, but it may extend to the phreatic zone. Although tracer studies in a British epikarst have shown that vertical flow velocities (to the caves below) may exceed 100 m/h (300 ft/h), dye was still detectable 13 months later [62–64]. In spite of the fact that most dye (and, presumably, most pollutants) travels through an aquifer at high velocities, some may become “hung up” in epikarstic storage. This flow dichotomy is actually a continuum, but it suggests that the common adage, that ground-water pollution in karst areas is not a long-term problem because the aquifer is rapidly self-cleaned, is wrong—or, at least, unreliable.

Exception 4: Fuzzy and Overlapping Basin Boundaries

It is quite logical to assume that a ground-water basin has a specific, well-defined boundary. Sometimes the boundary is breached and, during response to moderate- to flood-flow conditions, some water is diverted to adjacent ground-water basins, as already discussed. This assumption is reasonable and generally correct; it can be extended to allow for temporary, slight shifts in the boundary between two basins, also in response to storm-related changes in flow conditions. In contrast, some diffuse-flow aquifers show significant exceptions to this implicit assumption; the basin boundaries may be nebulous and gradational during all flow conditions. For example, all dye injected near the center of a ground-water basin in the Great Oolite Limestone (in the Bath district of southwestern England) flows

to only one spring. As different dyes from successive tests are injected at increasingly greater distances from the central axis of the basin, less dye goes to its spring. The balance of the dye goes to the major spring draining the adjacent ground-water basin. As the dye injection point gets successively farther from the central axis, more and more dye goes to the major spring in the adjacent basin. Such fuzzy boundaries are characteristic of basins in this aquifer. Smart has recommended that the gradational "boundary" between any two such ground-water basins be chosen to coincide with tracer injection springs [65]. Similar results occur elsewhere in England [66] and probably in other places where diffuse flow predominates. Nebulous, gradational boundaries between ground-water basins have recently been recognized in Missouri [24] and undoubtedly will be recognized in other karst terranes of the United States.

Gradational basin boundaries are suggested by the results of dye injection into a swallet (a sinkhole into which a stream empties) at the west boundary of the Turnhole Spring ground-water basin (Fig. 2, on the 520-ft contour). This swallet drains to two different ground-water basins: it flows both to the northeast (within the Turnhole Spring basin) and to the west, to a second major drainage basin (Ref 67, pages 48–49). This Kentucky boundary has not been studied, but I believe that the extent of overlap there is less than half a kilometre.

One should be prepared to encounter fuzzy and overlapping boundaries between ground-water basins. The possible existence of such boundaries makes it necessary to be extremely thorough in the design of dye tracing investigations and confirms the already recognized need for monitoring for dye at springs in ground-water basins adjacent to a proposed facility [27].

Overlap of ground-water basins is more convincingly illustrated by the Bear Wallow basin in Kentucky [12]. It occupies 500 km² (190 mile²) and its three subbasins, Hidden River, Three Springs, and Uno, resemble a Venn diagram (a diagram employed in symbolic logic; it uses circles and their relative position to represent sets and their relationships [68,69]). These subbasins occupy 65, 15, and 20%, respectively, of the total basin, but the Uno subbasin comprises part of the headwaters of the other two and is common to each of them. This overlap is significant because the 99 km² (38 mile²) of the Uno basin is the size of the area alluded to earlier in this paper as the terrane in Fig. 2, from which pollutants could flow to a total of 52 springs in 19 isolated segments along a 19-km (12-mile) reach of Green River. The probable consequences of leakage from a facility would be significantly fewer if it were located somewhere other than in a ground-water basin in which the headwaters are analogous to a Venn diagram and discharge is via distributary flow.

Exception 5: Noncontiguous Ground-Water Basins

Thraillkill and his students have shown that there are two physically distinct spring types in karst of the Inner Bluegrass of Kentucky: local high-level springs discharging from shallow flow paths and major low-level springs discharging from a deep, integrated conduit system [70–72]. The major springs are characterized by larger catchment areas [>10 km² (4 mile²)] and higher discharges [10 to 2700 L/s (2.6 to 700 gal/s)], in comparison with the smaller catchment areas of the local high-level springs [<2 km (0.8 mile²)] and their lower discharges [0.1 to 0.8 L/s (0.037 to 0.2 gal/s)]. The lack of integration of the local high-level spring catchments into the major low-level spring catchments can be explained in terms of the impermeability of numerous interbedded shales and the lack of fractures passing through them. The catchment of each major low-level ground-water basin is near-elliptical, unrelated to surface drainage, isolated from the nearby major basins, and commonly separated from its nearest similar neighbor by 1 to 4 km (0.6 to 2.5 mile). The reasons for

the noncontiguity of these major basins are not yet understood but may be related to the nature of the epikarst between conduits and the inhibition of hydraulic integration by clay and shale [73].

Such noncontiguous basins within a karst dipping uniformly at a low angle are not known to occur elsewhere, but they probably exist. The possibility of their occurrence in various settings should be anticipated and can be detected when traces are run from each side of a suspected boundary of a ground-water basin.

It is possible that traditional, randomly located wells may provide effective monitoring in the interbasin areas between noncontiguous basins. To use these wells reliably in such a karst, however, the tracing procedures advocated herein must first be employed and competently shown to yield negative results at springs and positive results at the wells.

Exception 6: Diversion of Ground-Water to the Surface

The Poorhouse Spring ground-water basin, in Kentucky, nicely illustrates another idiosyncrasy of flow in karst [12]. The basin drains 70 km² (27 mile²) and most flow is to the southeast, through the 3 by 6-m (10 by 20-ft) trunk stream passage that is Steele's Cave. About once a year, after a very heavy rain, some of the trunk's flow rises about 21 m (70 ft) above its normal level, flows out of the sinkhole entrance, and is discharged onto the surface. From there it flows in the opposite direction, to the northwest and west, where it augments the flow of a surface stream which drains an area outside of the Poorhouse Spring ground-water basin. Such flow, if contaminated, could give spurious values at surface sites used to monitor pollutants or background in adjacent ground-water and surface-water basins.

Exception 7: Diversion of Surface Water to a Different Ground-Water Basin

Sometimes surface waters are diverted during storms from one ground-water basin to another. This is illustrated by the behavior of Cayton Branch, in Kentucky (Refs 12 and 67, page 22). Little Sinking Creek (the southwesternmost surface stream in Fig. 2) drains north to the Green River, as shown, and contributes to the discharge from the Turnhole Spring ground-water basin. At a point about 2 km (1.2 miles) south of the creek's northernmost swallet, where the south fork of the stream bends north at the 600-ft potentiometric contour, the creek goes out of its banks during floods and diverts some of its discharge westward. The diverted surface water flows about 600 m (2000 ft) west to a swallet that comprises part of a larger ground-water basin in which the headwaters have been captured by the Turnhole Spring ground-water basin. Although the northward piracy of the northwest-flowing south fork of Little Sinking Creek seems to be a surface piracy, it is probably related to subsurface piracy of the adjacent western ground-water basin (and others) by the Turnhole Spring ground-water basin [74]. The present-day hydraulic gradient from its swallet, northward to Mill Hole (midway between it and Turnhole Spring), is about 9 m/km (47 ft/mile), approximately four times steeper than the present-day gradient westward to the spring along Barren River to which the south fork formerly flowed and to which the storm-diverted surface water now flows [12].

This diversion of surface water from the Turnhole Spring basin to another ground-water basin could be relevant to a monitoring effort in the second basin if there were significant quantities of pollutants in the surface waters of Little Sinking Creek and if they were "exported" to the second basin, where they could adversely affect the reliability of data from cave streams used for monitoring of background.

The above exceptions to the seven implicit assumptions stated as hydrologic axioms are

rare, but important. Their existence justifies a thoroughness in the design of tracer tests and in the interpretation of tracer results that, to some people, might seem almost paranoid. Their existence emphasizes the need for facility-related fieldwork. Additional exceptions are discussed in Ref 27.

Summary

In order to be relevant to monitoring for pollutants, water-quality data from karst terranes must be from springs, cave streams, and wells which have been shown by tracing to drain from the site to be monitored. Tracing should typically be done at least three times: when first convenient (during moderate-flow conditions, to quickly provide a preliminary, tentative understanding of local movement of ground water), and later, during base flow and flood flow. Sites for monitoring background should be selected on the basis of negative results of these tracing tests and should be in settings in which the rocks and waters are geochemically similar to those of the locations where the tracer tests were positive. The general prudence of tracing during all three types of flow conditions can not be overemphasized.

A map of the potentiometric surface, if it is based on enough data from an aquifer not complicated by aquicludes and aquitards, will greatly enhance one's ability to design the necessary tracing tests efficiently and to interpret them with greater confidence. One well-designed tracing test, properly done and correctly interpreted, is worth 1000 expert opinions—or 100 computer simulations of ground-water flow.

Sampling for water quality must be frequent and done before, during, and after storm and meltwater events. Base flow should be sampled between such events.

Ground-water monitoring in karst terranes can be done reliably, but the analytical costs are likely to be significantly higher than those for other terranes. It could be far less expensive to locate a proposed facility in a nonkarst terrane.

A plethora of ways exists to design the placement and sampling frequency of a ground-water monitoring network in a karst terrane so that it yields falsely negative results for the chemical compound being sought. Accordingly, environmental consultants and regulators must be ever vigilant to be sure that negative results are not falsely negative—either accidentally or intentionally.

Numerous plausible, axiomatic rules can be stated about ground-water flow in karst terranes, but they are not absolutes. Exceptions are known for each of the seven rules cited; more exceptions will be discovered.

Epilogomenon

A question can be raised of why professional geologists and engineers—who would not think of venturing into the design of a building foundation, a landfill, or a well field without first having obtained some initial experience with such matters or without a review of its design by a competent peer—all too often assume that anyone can do professional-quality dye tracing the first time it is attempted. If one's tracing experience is limited or nonexistent, the most astute, most ethical, and least expensive ways to minimize the risks of costly litigation for tracing-related malpractices are to accomplish the following tasks:

- (a) obtain experience with one of the few individuals in North America who are adept at dye tracing in the field;
- (b) hire one of these individuals to do the tracer investigation—with the understanding that he will provide some rudimentary training;

- (c) hire one of them to design the tracer investigation; or,
- (d) at an absolute minimum, hire one of them to review the design of the proposed tracer investigation.

An experienced, adept tracing consultant potentially saves his employer many times his fee—and greatly enhances the reliability of both the tracer results and the proposed monitoring system.

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References

- [1] Quinlan, J. F. and Ewers, R. O., "Ground Water Flow in Limestone Terranes: Strategy Rationale and Procedure for Reliable, Efficient Monitoring of Ground Water Quality in Karst Areas," *Proceedings, Fifth National Symposium and Exposition on Aquifer Restoration and Ground Water Monitoring*, Columbus, OH, National Water Well Association, Worthington, OH, 1985, pp. 197–234.
- [2] Smart, P. L. and Hobbs, S. L., "Characterization of Carbonate Aquifers: A Conceptual Base," *Proceedings, Conference on Environmental Problems in Karst Terranes and Their Solutions*, Bowling Green, KY, National Water Well Association, Dublin, OH, 1987, pp. 1–14.
- [3] Quinlan, J. F. and Alexander, E. C., Jr., "How Often Should Samples Be Taken at Relevant Locations for Reliable Monitoring of Pollutants from an Agricultural, Waste Disposal, or Spill Site in a Karst Terrane? A First Approximation," *Proceedings, Second Multidisciplinary Conference on Sinkholes and the Environmental Impacts of Karst*, Orlando, FL, Balkema, Rotterdam, 1987, pp. 277–286.
- [4] Crawford, N. C., "Sinkhole Flooding Associated with Urban Development upon Karstic Terrain: Bowling Green, Kentucky," *Proceedings, Multidisciplinary Conference on Sinkholes*, Orlando, FL, Balkema, Rotterdam, 1984, pp. 283–292.
- [5] Aley, T., "A Model for Relating Land Use and Groundwater Quality in Southern Missouri," *Hydrologic Problems in Karst Areas*, R. R. Dilamarter and S. C. Csallany, Eds., Western Kentucky University, Bowling Green, KY, 1977, pp. 323–332.
- [6] Aley, T. and Thompson, K. C., "Septic Fields and the Protection of Groundwater Quality in Greene County, Missouri, Final Report," consultant's report, Ozark Underground Laboratory, Protem, MO, 1984.
- [7] Quinlan, J. F., "Recommended Procedure for Responding to Spills of Hazardous Materials in Karst Terranes," *Proceedings, Conference on Environmental Problems in Karst Terranes and Their Solutions*, Bowling Green, KY, National Water Well Association, Dublin, OH, 1987, pp. 183–196.
- [8] Smoot, J. L., Mull, D. S., and Lieberman, T. D., "Quantitative Dye Tracing Techniques for Describing the Solute Transport Characteristics of Ground-Water Flow in Karst Terrane," *Pro-*

- ceedings, Second Multidisciplinary Conference on Sinkholes and the Environmental Impacts of Karst, Orlando, FL, Balkema, Rotterdam, 1987, pp. 269–275.
- [9] Bonacci, O., *Karst Hydrogeology, with Special Reference to the Dinaric Karst*, Springer-Verlag, New York, 1987.
 - [10] Milanović, P. T., *Karst Hydrogeology*, Water Resources Publications, Littleton, CO, 1979.
 - [11] White, W. B., *Geomorphology and Hydrology of Karst Terranes*, Oxford Press, New York, 1988.
 - [12] Quinlan, J. F. and Ray, J. A., "Groundwater Basins in the Mammoth Cave Region, Kentucky," *Friends of the Karst Occasional Publication*, No. 1, 1981.
 - [13] Palmer, A. N., "Prediction of Contaminant Paths in Karst Aquifers," *Proceedings, Conference on Environmental Problems in Karst Terranes and Their Solutions*, Bowling Green, KY, National Water Well Association, Dublin, OH, 1987, pp. 32–53.
 - [14] Lange, A. L., "Detection and Mapping of Karst Conduits from the Surface by Acoustic and Natural Potential Methods," research report prepared for the National Park Service and U.S. Environmental Protection Agency, The Geophysics Group, Wheat Ridge, CO, 1988.
 - [15] Stokowski, S. J., Jr., "Locating Groundwater Conduits in Carbonate Rocks," *Proceedings, Multidisciplinary Conference on Sinkholes and the Environmental Impacts of Karst*, Orlando, FL, Balkema, Rotterdam, 1987, pp. 185–196.
 - [16] Werner, E., "Photolineaments and Karst Conduit Springs in the Greenbrier Limestones of West Virginia," Abstract, *Geological Society of America Abstracts with Programs*, Vol. 20, 1988, p. 322.
 - [17] Parizek, R. R., "On the Nature and Significance of Fracture Traces and Lineaments in Carbonate and Other Terranes," *Karst Hydrology and Water Resources*, Vol. 1, V. Yevjevich, Ed., Water Resources Publications, Fort Collins, CO, 1976, pp. 47–100.
 - [18] Quinlan, J. F., Ewers, R. O., and Field, M. S., "How to Use Ground-Water Tracing to 'Prove' that Leakage of Harmful Materials from a Site in a Karst Terrane Will Not Occur," *Proceedings, Second Conference on Environmental Problems in Karst Terranes and Their Solutions*, Nashville, TN, National Water Well Association, Dublin, OH, 1988, pp. 289–301.
 - [19] Smart, C. C. and Ford, D. C., "Structure and Function of a Conduit Aquifer," *Canadian Journal of Earth Sciences*, Vol. 23, 1986, pp. 919–929.
 - [20] Smart, C. C. "Quantitative Tracing of the Maligne Karst System, Alberta, Canada," *Journal of Hydrology*, Vol. 98, 1988, pp. 185–204.
 - [21] Smart, C. C. "Quantitative Tracer Tests for the Determination of the Structure of Conduit Aquifers," *Ground Water*, Vol. 26, 1988, pp. 445–453.
 - [22] Esposito, A., *Fluid Power with Applications*, Prentice-Hall, Englewood Cliffs, NJ, 1980.
 - [23] Quinlan, J. F. and Rowe, D. R., "Hydrology and Water Quality in the Central Kentucky Karst: Phase I," *University of Kentucky Water Resources Research Institute, Research Reports*, No. 101, 1976; reprinted 1977, with corrections, as Uplands Field Research Laboratory (National Park Service) Management Report No. 12.
 - [24] Aley, T., "Complex Radial Flow of Ground Water in Flat-Lying Residuum-Mantled Limestone of the Arkansas Ozarks," *Proceedings, Second Conference on Environmental Problems of Karst Terranes and Their Solutions*, Nashville, TN, National Water Well Association, Dublin, OH, pp. 159–170.
 - [25] Quinlan, J. F., "Ground-Water Monitoring in Karst Terranes: Recommended Protocols and Implicit Assumptions," EPA/600/X-89/050, U.S. Environmental Protection Agency, Environmental Monitoring Systems Laboratory, Las Vegas, NV, 1989.
 - [26] Beck, B. F., Quinlan, J. F., and Ewers, R. O., "Presentation of the E. B. Burwell, Jr., Memorial Award to James F. Quinlan and Ralph O. Ewers," *Geological Society of America Bulletin*, Vol. 97, 1987, pp. 141–143.
 - [27] Aley, T. J., Quinlan, J. F., and Behrens, H., *The Joy of Dyeing: A Compendium of Practical Techniques for Tracing Groundwater, Especially in Karst Terranes*, National Water Well Association, Dublin, OH, in press.
 - [28] Libra, R. D., Hallberg, G. R., Hoyer, B. E., and Johnson, L. G., "Agricultural Impacts on Ground Water Quality: The Big Spring Basin Study, Iowa," *Proceedings, Agricultural Impacts on Ground Water*, Omaha, NE, National Water Well Association, Dublin, OH, 1986, pp. 253–273.
 - [29] Hallberg, G. R., Libra, R. D., and Hoyer, B. E., "Nonpoint Source Contamination of Ground Water in Karst-Carbonate Aquifers in Iowa," *Perspectives in Nonpoint Source Pollution*, EPA 440/5-85-001, U.S. Environmental Protection Agency, Washington, DC, 1985, pp. 109–114.
 - [30] Hoyer, B. E., Iowa Geological Survey, personal communication, December 1987.
 - [31] "Atrazine Health Advisory" (draft), *Basic Documents, National Pesticide Survey*, U.S. Environmental Protection Agency, Office of Drinking Water, Washington, DC, 1987.
 - [32] Sanders, T. G., Ward, R. C., Loftis, J. S., Steele, T. D., Adrian, D. D., and Yevjevich, V., *Design of Networks for Monitoring Water Quality*, Water Resources Publications, Littleton, CO, 1983.

- [33] Gilbert, R. O., *Statistical Methods for Environmental Pollution Monitoring*, Van Nostrand Reinhold, New York, 1987.
- [34] Makridakis, S., Wheelwright, S. C., and McGee, V. E., *Forecasting: Methods and Applications*, 2nd ed., Wiley, New York, 1983.
- [35] Chatfield, C., *The Analysis of Time Series: An Introduction*, 3rd ed., Chapman and Hall, London, 1984.
- [36] Bendat, J. S. and Piersol, A. G., *Random Data: Analysis and Measurement Procedures*, 2nd ed., Wiley, New York, 1986.
- [37] Gibbons, R. D., "Statistical Prediction Intervals for the Evaluation of Ground-Water Quality," *Ground Water*, Vol. 25, 1987, pp. 455-465.
- [38] Montgomery, R. H., Loftis, J. C., and Harris, J. "Statistical Characteristics of Ground-Water Quality Variables," *Ground Water*, Vol. 25, 1987, pp. 176-184.
- [39] Beck, M. B. and van Straten, G., Eds., *Uncertainty and Forecasting of Water Quality*, Springer-Verlag, Berlin, 1983.
- [40] Quinlan, J. F. and Aley, T., "Discussion of 'A New Approach to the Disposal of Solid Waste on Land,'" *Ground Water*, Vol. 25, 1987, pp. 615-616.
- [41] Quinlan, J. F., "Hydrologic Research Techniques and Instrumentation Used in the Mammoth Cave Region, Kentucky," *1981 GSA Cincinnati '81 Field Trip Guidebooks*, Vol. 3, T. G. Roberts, Ed., American Geological Institute, Washington, DC, 1981, pp. 502-504.
- [42] Quinlan, J. F., "Groundwater Basin Delineation with Dye-Tracing, Potentiometric Surface Mapping, and Cave Mapping, Mammoth Cave Region, Kentucky, U.S.A.," *Beitrage zur Geologie der Schweiz—Hydrologie*, Vol. 28, 1982, pp. 177-189.
- [43] Smart, P. L. "A Review of the Toxicity of Twelve Fluorescent Dyes Used in Water Tracing," *National Speleological Society Bulletin*, Vol. 46, pp. 21-33.
- [44] Thrailkill, J., Byrd, P. E., Sullivan, S. B., Spangler, L. E., Taylor, C. J., Nelson, G. K., and Pogue, K. R., "Studies in Dye-Tracing Techniques and Karst Hydrogeology," *University of Kentucky, Water Resources Research Institute, Research Report*, No. 140, 1983.
- [45] Duley, J. W., "Water Tracing Using a Scanning Spectrofluorometer for Detection of Fluorescent Dyes," *Proceedings*, Conference on Environmental Problems in Karst Terranes and Their Solutions, Bowling Green, KY, National Water Well Association, Dublin, OH, 1987, pp. 389-406.
- [46] Molz, F. J., Guven, O., Melville, J. G., Crocker, R. D., and Matteson, K. T., "Performance, Analysis, and Simulation of a Two-Well Tracer Test," *Water Resources Research*, Vol. 22, 1986, pp. 1031-1037.
- [47] Fetter, C. W., *Applied Hydrogeology*, 2nd ed., Merrill, New York, 1988.
- [48] Winters, S. L. and Lee, D. R., "In Situ Retardation of Trace Organics in Groundwater Discharge to a Sandy Stream Bed," *Environmental Science and Technology*, Vol. 21, 1987, pp. 1182-1186.
- [49] Tóth, J., "A Theory of Ground-Water Motion in Small Drainage Basins in Central Alberta, Canada," *Journal of Geophysical Research*, Vol. 67, 1962, pp. 4375-4387.
- [50] Freeze, R. A. and Witherspoon, P. A., "Theoretical Analysis of Regional Groundwater Flow," *Water Resources Research*, Vol. 3, 1967, pp. 623-634.
- [51] Wiersman, J. H., Stieglitz, R. D., Cecil, D. L., and Metzler, G. M., "Characterization of the Shallow Groundwater System in an Area with Thin Soils and Sinkholes," *Proceedings*, First Multidisciplinary Conference on Sinkholes, Orlando, FL, Balkema, Rotterdam, 1984, pp. 305-310.
- [52] Bradbury, K. R., "Hydrogeologic Relationships Between Green Bay of Lake Michigan and Onshore Aquifers in Door County, Wisconsin," Ph.D. thesis, University of Wisconsin-Madison, Madison, WI, 1982.
- [53] Cherkauer, D. S., Taylor, R. W., and Bradbury, K. R., "Relation of Lake Bed Leakage to Geoelectrical Properties," *Ground Water*, Vol. 25, 1987, pp. 135-140.
- [54] Keagy, D., Ewers, R. O., and Quinlan, J. F., studies in progress, 1988.
- [55] Lindholm, G. F., "Snake River Plain Regional Aquifer-System Study," *U.S. Geological Survey Circular*, No. 1002, 1986, pp. 88-106.
- [56] Von Knebel, W., *Hohlenkunde mit Berücksichtigung der Karst-phenomene*, Viewig, Braunschweig, Germany, 1906.
- [57] Maclay, R. W. and Small, T. A., "Carbonate Geology and Hydrology of the Edwards Aquifer in the San Antonio Area, Texas," *U.S. Geological Survey Open-File Reports*, No. 83-537, 1984.
- [58] Campana, M. E. and Mahin, D. A., "Model-Derived Estimates of Groundwater Mean Ages, Recharge Rates, Effective Porosities and Storage in a Limestone Aquifer," *Journal of Hydrology*, Vol. 76, 1985, pp. 247-264.
- [59] Miller, J. A., "Hydrogeologic Framework of the Floridan Aquifer System in Florida and in Parts of Georgia, South Carolina, and Alabama," *U.S. Geological Survey Professional Papers*, No. 1403-B, 1984.

- [60] Beck, B. F., "Ground Water Monitoring Considerations in Karst on Young Limestones," *Proceedings*, Conference on Environmental Problems in Karst Terranes and Their Solutions, Bowling Green, KY, National Water Well Association, Dublin, OH, 1987, pp. 229-248.
- [61] Williams, P. W., "Subcutaneous Hydrology and the Development of Doline and Cockpit Karst," *Zeitschrift für Geomorphologie*, Vol. 29, 1985, pp. 463-482.
- [62] Friederich, H. and Smart, P. L., "Dye Tracer Studies of the Unsaturated-Zone Recharge of the Carboniferous Limestone Aquifer of the Mendip Hills, England," *Proceedings*, Eighth International Speleological Congress, Vol. 1, Bowling Green, KY, 1981, pp. 283-286.
- [63] Friederich, H. and Smart, P. L., "The Classification of Autogenic Percolation Waters in Karst Aquifers: A Study in GB Cave, Mendip Hills, England," *University of Bristol Speleological Society Proceedings*, Vol. 16, 1982, pp. 143-159.
- [64] Smart, P. L. and Friederich, H., "Water Movement in the Unsaturated Zone of a Maturely Karstified Carbonate Aquifer, Mendip Hills, England," *Proceedings*, Conference on Environmental Problems in Karst Terranes and Their Solutions, Bowling Green, KY, National Water Well Association, Dublin, OH, 1987, pp. 59-87.
- [65] Smart, P. L., "Catchment Delimitation in Karst Areas by Use of Quantitative Tracer Methods," *Proceedings*, Vol. 2, Third International Symposium on Underground Water Tracing, Ljubljana-Bled, Yugoslavia, 1976, pp. 291-298.
- [66] Atkinson, T. C. and Smart, P. L., "Artificial Tracers in Hydrogeology," *A Survey of British Hydrogeology*, The Royal Society, London, 1981, pp. 173-190.
- [67] Quinlan, J. F., Ewers, R. O., Ray, J. A., Powell, R. L., and Krothe, N. C., "Groundwater Hydrology and Geomorphology of the Mammoth Cave Region, Kentucky, and of the Mitchell Plain, Indiana," *Field Trips in Midwestern Geology*, Vol. 2, R. H. Shaver and J. A. Sunderman, Eds., Geological Society of America and Indiana Geological Survey, Bloomington, IN, 1983, pp. 1-85.
- [68] Gardner, M., *Logic Machines and Diagrams*, 2nd ed., University of Chicago Press, Chicago, 1982, pp. 32, 39-54.
- [69] Miller, R. W., *Study Guide, "Introduction to Logic,"* 7th ed., Macmillan, New York, 1986, pp. 75-76, 79, 83-96.
- [70] Scanlon, B. R. and Thrailkill, J., "Chemical Similarities Among Physically Distinct Spring Types in a Karst Terrain," *Journal of Hydrology*, Vol. 89, 1987, pp. 259-279.
- [71] Thrailkill, J., "Flow in a Limestone Aquifer as Determined from Water Tracing and Water Levels in Wells," *Journal of Hydrology*, Vol. 78, 1985, pp. 123-136.
- [72] Thrailkill, J., "Hydrogeology and Environmental Geology of the Inner Bluegrass Karst Region, Kentucky," (Field Guide), Annual Meeting, Geological Society of America, Southeastern and North-Central Sections, Lexington, KY, 1984.
- [73] Ewers, R. O., Department of Geology, Eastern Kentucky University, personal communication, January 1988.
- [74] Quinlan, J. F. and Ewers, R. O., "Preliminary Speculations on the Evolution of Groundwater Basins in the Mammoth Cave Region, Kentucky," *GSA Cincinnati '81 Field Trip Guidebooks*, Vol. 3, T. G. Roberts, Ed., American Geological Institute, Washington, DC, 1981, pp. 496-501.

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